EFFECTS OF THINNING ON FOREST STRUCTURE AND COMPOSITION IN THE WUNGONG CATCHMENT, WESTERN AUSTRALIA

EFFECTS OF THINNING ON FOREST STRUCTURE AND COMPOSITION IN THE WUNGONG CATCHMENT, WESTERN AUSTRALIA

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Declaration

I declare that this thesis is my own account of my research and contains as its main content work which has not previously been submitted for a degree at any tertiary educational institution.

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Sita Ram Panta

Abstract

The focus of jarrah (*Eucalyptus marginata* Donn ex. Smith) forest management in past decades has been changing in response to forest management practices of the day, interests and needs of the forest users and more recently a changing climate. Following uncontrolled logging including clear-felling and wild-fires after European settlement (1830s onwards) and rotational logging and burning under the state forest management plan, much of the old-growth forest of south-west Western Australia has been converted into a dense re-growth forest. Substantial reduction in the rainfall in the south-west of Western Australia in the past three decades has resulted in a significant decrease in water available to the Integrated Water Supply Scheme at a time of increasing water demand of the region. A forest thinning trial to increase both water yield and environmental benefits from the catchment has been implemented since 2006 in the Wungong Catchment within state forest. Wungong forest is now managed by the Water Corporation as a drinking water catchment in association with the Department of Environment and Conservation (DEC) and Forest Product Commission (FPC); and is under the scrutiny of Environmental Protection Authority (EPA) of Western Australia. Although the effects of forest thinning on catchment hydrology are well understood, short-term effects of thinning on forest ecosystems especially on forest structure and composition are less well understood. The research reported in this thesis contributes to understanding the short-term effects of thinning on stand structure and on the composition of overstorey and understorey components of the forest, including the dead woody debris (DWD).

Two thinning strategies, i.e., commercial logging followed by glyphosate herbicide treatment of selected non-commercial trees (log+notch) or killing of selected trees by applying glyphosate herbicide to notches in the trunk (notch-only), were compared as treatments to the adjacent unthinned stands (control) in the Wungong Catchment within jarrah forest of south-

west Western Australia. All three treatments were replicated three times using plots of 90 x 70 m size. Thinning, which was carried out in September, 2009, reduced tree basal area from 50.2 to 17.9 m^2 ha^{-1} for log+notch and 43.8 to 22.2 m^2 ha^{-1} for notch-only treatment.

Forest structure variables were investigated in relation to thinning *viz*: stand density and basal area for three dominant tree species and their composition in the overstorey; understorey species richness and ground cover; and quantity, structure and composition of DWD.

This study revealed that both log+notch and notch-only thinning treatments significantly reduced the overall stand density and basal area and changed the composition of the treated tree stands compared to the unthinned stands. Frequency distributions of the tree stands in different size classes were altered with highly significant decreases in 20-30 and 30-40 cm diameter at breast height (DBH) classes. By contrast, no significant effect was observed in the understorey species richness and ground cover within 1 year after thinning, partly due to the absence of direct physical disturbance in the sampled quadrats. However, there was a slight reduction in thinned as well as control forest in the species richness recorded during the hot and dry season compared to the winter sampling. Results suggest that selection of sampling units representing different micro-site conditions within the study area are important and the sampling area needs to be increased to 64 m^2 (16 lots of 2x2 m quadrats) to adequately represent species richness present in the study area.

Thinning increased the quantity of DWD by 100 % in log+notch and 142 % in notch-only treatments. Thinning also changed the structure and composition of DWD components (i.e. log, snag and stump) by altering the percentage contribution of each component to the total DWD pool. Thinning altered the percentage contribution of log, snag and stump components to the total DWD volume from 80, 17 and 3 % before thinning to 48, 50 and 2 %, respectively, for log+notch treatment after 1 year, and from 86, 11 and 2 % to 36, 63 and 1 %,

respectively, for notch-only treatments. Changes in the structure of the dead woody resources after thinning revealed that the amount, structure and composition of DWD in the forested catchment was directly related to stand age and structure of the living trees and more specifically to the method and intensity of forest thinning.

In summary, this study suggests that thinning as a management intervention is an important driver of the structure and composition dynamics of jarrah forest ecosystems. The level of reduction in the stand density and basal area of the tree stand and increase in the amount of DWD as compared to the adjacent control stands were dependent on the methods and intensity of thinning. Although the effects of thinning on overstorey components such as species composition, stand density and diameter size-class distribution of tree stands, and on DWD components were significant, effects on understorey species richness and ground cover were not discernable in the short-term. Hence, longer term study of post-thinning vegetation structure and composition are needed to adequately describe thinning impacts on post-thinning vegetation dynamics and successional processes.

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Abbreviations and Symbols

DEC	Department of Environment and Conservation
DWD	Dead Woody Debris
CWD	Coarse Woody Debris
FPC	Forest Products Commission
CALM	Conservation and Land Management
CC	Conservation Commission
EPA	Environmental Protection Authority
FDWA	Forest Department of Western Australia
WA	Western Australia
USA	United States of America
USDA	United States Department of Agriculture
DBH	diameter at breast height (1.37 m)
TDM	total dry mass
BA	basal area
LAI	leaf area index
NA	not applicable
ha	hectare

mm	millimetre
cm	centimetre
m	metre
m^2	square metre
ha^{-1}	per hectare
$year^{-1}$	per year
˚C	degree centigrade
g	gram
l	litre
t	tonne
kg	Kilogram
ln	natural log
>	more than
<	less than
≥	more and or equal to
≤	less and or equal to
L	Log+notch treatment
C	Control treatment
N	Notch-only treatment

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Dedication

Dedicated to my late father

Chhabi Lal Panta

Chapter 1: Introduction

1.1 The Jarrah forest

Prior to settlement by Europeans, Jarrah (*Eucalyptus marginata* Donn ex. Smith) forest was distributed throughout south-western Australia covering over 5.3×10^6 ha. However, clearing of the land for agriculture has reduced the area of the Jarrah forest to about 3.3×10^6 ha, mostly on crown land (Dell and Havel 1989). More recent data suggest that the jarrah forests occupy about 1.57 million hectares of the south-western corner of Western Australia (Whitford and Williams 2001). They occur primarily on the highly leached and weathered lateritic soils of the Darling Plateau which forms the south-western and higher-rainfall part of the Great Plateau of Western Australia (Jutson 1934). The laterite profile has developed by *in situ* deep weathering. Average annual rainfall ranges from 750 to 1400 mm, with prime forest in areas where rainfall exceeds 1000 mm. The Jarrah forest extends from Toodyay in the north to Albany in the south and its eastern limit is approximately the 635 mm isohyet (Gentilli 1989).

On the basis of canopy density and height, jarrah forest is classified as open forest in its northern range and as tall forest in its southern range (Specht et al. 1974). As there is a reduction in rainfall distribution towards the north and east, the trees decrease in stature thus changing the vegetation form of Jarrah forest in the drier reaches of its distribution to woodland or low forest. This dry sclerophyll forest is dominated by jarrah, followed by marri (*Corymbia calophylla* Lindl.) in the overstorey. There is an understorey of small tree species of about 4-7 m height, mainly sheoak (*Allocasuarina fraseriana* Miq.), bull banksia (*Banksia grandis* Willd.), snottygobble (*Persoonia longifolia* R. Br.), spreading snootygobble (*Persoonia elliptica* R. Br.), and a ground cover of woody shrubs comprising the grass-tree (*Xanthorrhoea preissii* Endl.), kingia (*Kingia australis* R. Br.), and a prominent cycad

(*Macrozamia riedlei* Fisch. ex Gaud.) and a range of other species predominantly from the families Anthericaceae, Dasypogonaceae, Leguminaceae, Orchidaceae, Apiaceae, Epacridaceae, Asteraceae, Proteaceae, Restionaceae, Myrtaceae and Cyperaceae (Green 1985). The northern jarrah forest was classified by previous authors into 21 site vegetation types based on a set of overstorey and understorey indicator species (Havel 1975a, Bell and Heddle 1989). These not only covered jarrah and marri-dominated forest communities that are typical of site vegetation types P, S and T but also the low open woodlands, open heaths and sedge lands of the stream zones and swamps for example, types A and C (Bell and Heddle 1989). This "Havel" site vegetation type classification is the most widely used mapping system in the northern jarrah forest (Bell and Heddle 1989).

The forests of the southwest of Western Australia enjoy a world-wide recognition for their exceptionally high plant diversity and are part of an international plant biodiversity hotspot (Myers et al. 2000). In contrast to most of the world's forests, the plant diversity of jarrah forest is mostly found in the understorey vegetation. The flora of the region is estimated to contain at least 784 species; however, the entire northern jarrah forest only contains approximately 20 indigenous species (Bell and Heddle 1989). This estimate includes swamps, streams and rocky outcrop habitats.

The distribution and floristic composition of jarrah forest have been influenced by several environmental factors; however, four major environmental factors influencing the vegetation of the jarrah forest region have been described (Havel 1979). These factors include topography of the region, available moisture, degree of soil leaching and the resulting acidity and the fertility of the soil. The interaction of all these factors along with the heritage of past forest use and management explain to a greater extent the present vegetation distribution and its floristic composition (Bell and Heddle 1989).

The vegetation characteristics of jarrah forest sites, especially floristic composition, are difficult to enumerate at present due to a large number of undescribed species (Hopper and Muir 1984). Moreover, the composition of particular jarrah forest sites fluctuates largely depending on the time since the last fire (Dell and Havel 1989), the impacts of *Phytophthora cinnamomi* Rands (Dell and Malajczuk 1989), alterations in the drainage patterns (Schofield et al. 1989), and other types of disturbance.

A total of approximately vascular 18,000 species are present in Australia, out of which Western Australia harbours 7,963 species contained in 1,226 genera and 181 families (Green 1985). There is uncertainty about the full list of flora of the region occurring only in the jarrah forest, but an approximation was prepared from the "Flora of the Perth Region" (Marchant et al. 1987). An accounting of the species was made based on the description for the location as being within the confines of the jarrah forest such as "Darling Range" or "Scarp" or indications of predominance in lateritic soils. Using these general guidelines, the jarrah forest contains approximately 784 species with the most common families being the Proteaceae, Papilionaceae and Myrtaceae with 70, 68 and 63 species, respectively. However, floristic richness is greatest in the sites that simultaneously contain many short-lived obligatory re-seeding species which are stimulated to germinate by high soil temperatures in a fire, the long-lived re-sprouting species commonly dominating the understorey and a few species which require dispersal from adjacent unburnt sites (Bell and Koch 1980).

1.2 Research background

Periodic logging and clear-felling of forest stands by the early millers in most of the south-west Australian forest for timber and fuel with little or no control, and the subsequent wildfires, have converted the old-growth forest to a dense re-growth forest (Water Corporation 2005). As re-growth forests are dense, they compete for the natural resources such as water, space, light and nutrients. A gradual conversion of heterogeneous old-growth forest into a dense homogenous re-growth forest and a dramatic decline in the rainfall across much of south-west Australia have led to a significant reduction in stream flows due to decline in surface run-off from the catchments.

The stream flow in the Wungong catchment has been reduced from 28.6 gigalitre (GL) per annum in 1911-1974 to 13.6 GL per annum in 1997-2003 resulting in reduced water levels in the Wungong dam (Water Corporation 2005). Recent work by Department of Environment and Conservation (DEC) and others on Stirling Dam Catchments predicts that a further 11 percent decline in rainfall over the next thirty years could likely result in a further 31 percent reduction in annual water yield (Berti et al. 2004). Reduced rainfall in the past decades and the predicted future climatic uncertainty have added a sense of urgency in West Australia to the search for new water resources to sustain the ever-growing population (Water Corporation 2005).

Building on the results of past thinning research conducted in Hansens, Higgens and Jones catchments (Water and Rivers Commission 1997) and in jarrah forest (Stoneman 1993), the Wungong Catchment Environment and Water Management Project (WCEWMP) proposed by the Water Corporation is in operation aiming to substantially increase water yield from the catchment (Water Corporation 2005). In this project, forest thinning is proposed as an attractive low-cost option for increasing water output from the catchment for the Perth metropolitan area and is potentially appropriate for other drinking water catchments. However, potential environmental and ecological impacts, which were the major community concerns, are still uncertain (Water Corporation 2005). To fulfil community expectations, the water yield from thinning must be sustainable and of high quality, with minimal effects on the catchment ecosystem as the thinned stands mature. In this regards, as part of the project “Balancing Water Quality and Ecosystem Health with Water Yield- Ecosystem Response to

Thinning in Wungong Catchment," this thesis is designed to assess the impacts of thinning on forest structure and composition within the Wungong catchment area. It is also a contribution to the Wungong Catchment Environment and Water Management Project (WCEWMP) which seeks to assess the potential effects of thinning on Jarrah forest ecosystems.

1.3 Research issues and questions

The value of any forested catchment keeps changing in response to factors such as climate change, forest management practices of the day and other activities of users (Water Corporation 2005). These activities while intended to fulfil the requirements of the day have significant effects on land-use pattern. As a result, there have been significant changes within the Wungong catchment through time (Bari and Ruprecht 2003). Development of dense regrowth forest after heavy cutting and some clear-felling with no or little control means they compete for limiting natural resources specifically the water and nutrients, led to the reduced water yield from the catchment (Water Corporation 2005). Roberts *et al.* (2000) indicate that the reductions in water yield averaging 30% were recorded for *Eucalyptus sieberi* regrowth stands aged between 8 and 20 years compared to previous mature stands in New South Wales. Similar studies in Victoria with 20 to 40-year-old regrowth stands of *Eucalyptus regnans* showed a reduction in water yield of between 50 and 60 % during the regrowth phase. Reduction in the stream flows are not only attributed to the growth and regeneration of the forests stands but also to the continuous reduction in the rainfall in the past years.

Although there is a significant increase in water output from the catchment after thinning, ecological impacts of forest thinning-which are the major community-concerns warrant further investigation. Also, there is insufficient research work on the effects of thinning on forest structure and composition especially in relation to water as the major limiting natural resource for jarrah forest growth.

1.4 Thesis aims and objectives

The present study was focussed on Wungong catchment, which is situated in the western margins of the Darling Plateau, an hour's drive south-east of Perth in Western Australia (WA). It is a drinking water catchment managed by the Water Corporation of Australia in association with DEC with timber harvesting rights available to FPC. This catchment management project is under the scrutiny of Environmental Protection Authority (EPA) of Western Australia. Forest thinning is proposed as a low cost adaptive catchment management option to increase sustainable water and environmental outputs from the existing catchment (Water Corporation 2005). However, the impacts of forest thinning on jarrah forest ecosystems are still unclear. This research was primarily aimed at assessing the response of a jarrah forest ecosystem to tree thinning. More specifically, objectives of this study were to:

1. Assess the structure and composition of the dominant forest tree species in response to thinning

2. Assess the effects of overstorey thinning on understorey species richness and ground cover

3. Assess the amount and structural composition of DWD in relation to forest management practices and thinning.

These three objectives lead to the following hypothesis:

Hypothesis 1: Log+notch and notch-only thinning treatments alter forest structure according to the prescribed decrease in density and basal area of smaller, high-water-use trees

Hypothesis 2: Log+notch and notch-only thinning treatments will produce different effects on understorey richness and ground cover

Hypothesis 3: Log+notch and notch-only thinning treatments will lead to a significant increase in the DWD but alter the distribution of DWD among types

While hypotheses 1 and 2 may require many years to test, only the short-term responses, in the year following thinning, were studied in the present thesis.

Chapter 2: Review of literature

Tree thinning has multiple effects on forest ecosystems (Borg et al. 1988). As thinning involves cutting or killing of standing trees, it is aimed primarily at controlling the growth of stands by adjusting stand density, stand size class distribution or species composition (Cochran and Barrett 1998). The general objective of thinning is to increase growth and improve stand health. However, vegetation structure, species richness, promotion of biodiversity and sustainable ecological balance are also of major concern (Peacock 2008). The basic assumption when applying a thinning treatment to a forest stand is that the growing space on a particular site has a limitation of available resources such as water, light and other inorganic nutrients (Brown 1997). Hence, it is assumed that thinning will promote the growth of desirable trees already present in the stand by releasing and re-allocating a greater proportion of those limiting resources to the retained plants (Brown 1997, Waddell 2002).

Forest thinning is not a new practice. Thinning from a forest improvement perspective has been practised for more than 100 years in the south-west of Western Australia (Bari and Ruprecht 2003), but it also has a long history in Canadian and North American forestry (Metzger and Schultz 1984, Alaback and Herman 1988). A review of previous studies suggests that thinning practices have varied considerably according to location, forest quality, objectives of the thinning and the forest management practices of the day and have produced diverse effects on different aspects of forest ecosystems (Borg et al. 1988). Different thinning studies conducted at different locations have produced location-specific results. Most thinning studies have reported a positive response of forest components such as increased growth rate of the retained trees (Stoneman et al. 1996, Brissette et al. 1999, Pothier 2002, Sullivan et al. 2002) while others have reported no response (Graae and Heskjaer 1997). Similarly, thinning has been reported to have significant effects on understorey species diversity and richness (Wienk et al. 2004, Metlen and Fiedler 2006). A thinning study conducted in jarrah forest recorded substantial increase in catchment water yield after thinning (Stoneman 1993); however other studies in the south-west of Western Australia demonstrated only a transient increase in water yield since water output returns to the pre-disturbance level after 12-15 years during the recovery of vegetation (Bari and Ruprecht 2003).

Disturbance whether it is natural or human induced, is fundamental to the development of forest ecosystems. Hence the management of natural areas should be based on an understanding of the disturbance processes (Attiwill 1994, Lorimer and Frelich 1994). Disturbance ecology encompasses the study of inter-relationships between biotic and abiotic components of the environment and has been variously defined according to the aims and objectives of the studies. A commonly quoted definition of disturbance is: " Any relatively discrete event in time that disrupts ecosystems, community, or population structure and changes resources, substrate availability, or the physical environment" (White and Pickett 1985). More recently as Federal land management agencies of the United States adopted ecosystem management philosophies, it became critical that disturbance was viewed as complementary to, rather than as solely deleterious to, human and forest functions (Monning and Byler 1992, Grumbine 1994).

Several authors have classified disturbances as either internal (endogenous) or external (exogenous) to the ecosystem; they can be biotic such as insects/pests, diseases and animal damage; or abiotic such as fire, flood, wind, drought and volcanic activity. The scale of disturbances may be small or large; intense or weak; and normal or devastating. One thing that most disturbance agents have in common, however, is that they rarely act alone. Some authors have linked drought to incidents of insect infestation (Mitchell et al. 1983, Hadley and Veblen 1993), disease (Baker 1988, Castello et al. 1995) and fire (Arno 1980, Romme and Despain 1989, Johnson and Larsen 1991, Callaway and Davis 1993). Winds have also been reported in several studies as disturbance agents that usually thin forests of trees already damaged by fire (Runkle 1985, Lorimer and Frelich 1994, Castello et al. 1995) and increase severity of fires known as "fire storms" burning large areas of forest in a single day (Romme and Despain 1989). Geomorphology and gravity also have significant effects on natural ecosystems through earth-shaping processes such as volcanism, earthquakes (Veblen et al. 1992) and glacial movements. In general, these disturbances tend to have long lasting effects because they often involve removal or destruction of the upper layer of soil horizons resulting into loss of biota from the landscapes that may take many years to re-establish. Several authors have looked at a variety of geomorphic disturbances where the key factor influencing potential disturbance energy was topography (Swanson et al. 1992).

The fragmentation of natural ecosystems by both natural and anthropogenic disturbances is one of the major threats to biodiversity worldwide. The fragmentation process enhanced by both natural and human activities reduces the area of native ecosystem consequently leaving isolated patches of native vegetation which are subject to species loss due to altered resource availability, invasion by non-native species and changes in population dynamics and ecosystem processes (Shafer 1990, Saunders et al. 1991, Bierregaad et al. 1992, Hobbs 1993). Most research on fragmented systems has focussed on species and, to a lesser extent, ecosystems (Yates et al. 1994), but there is an increasing recognition that landscape-level processes, particularly disturbances, are equally important for seedling regeneration of many plants species (Baker 1973, Grubb 1977, Pickett and Thompson 1978, Swanson et al. 1990, Baker 1992). Most of the Australian studies on plant species and community response to landscape-scale disturbances have focussed on fire (Gill et al. 1981, Bell et al. 1984, Ford 1985). It has been demonstrated that fire is essential for successful regeneration of many *Eucalyptus* species, and seedling recruitment in inter-fire periods is rare (Ashton 1976, O'Dowd and Gill 1984, Wellington and Noble 1985a, Wellington and Noble 1985b, Burrows et al. 1990, Hopper 1993). However, large-scale insect infestations and their impact on successional development of forest ecosystems have also been reported. These infestations may be associated with drought in the case of wood borers such as the mountain pine beetle (Ammon 1978, Mitchell et al. 1983) or sudden outbreak of the jarrah leaf miner in some parts of the jarrah forests (*Eucalyptus marginata*) in Western Australia over several decades (Mazanec 1974, 1985, 1989). Several other studies have reported some kind of insect pest outbreak causing high levels of defoliation in Australian eucalypt forests that have significant effects on post infestation successional processes (Campbell 1960, Greaves 1966, Carne et al. 1974, Springett 1978). Similar is the occurrence of dieback disease in the Western Australian forest, particularly in the cut-over jarrah forest near roadways and forestry settlements (Podger 1972, Batini 1973, 1974). Moreover, landscape-scale disturbances in Australia such as drought (Pook et al. 1966, Hnatiuk and Hopkins 1980), flood (Bren 1993) and severe storms (Minor et al. 1980, Unwin et al. 1988) have also been reported to occur.

Landscape-scale disturbances and regeneration studies conducted in semi-arid woodlands of South-Western Australia have examined the responses of unfragmented woodlands to disturbances caused by fire, floods, windstorms and drought (Yates et al. 1994). Sites known to have any of several disturbance types over the previous 50 years or so had been reported to have cohorts of sapling-stage *Eucalyptus salmonophloia* F. Muell. (salmon gum) and other dominant *Eucalyptus* species (Yates et al. 1994). Sites disturbed either by fire, flood or storms during 1991-1992 displayed adult tree mortality and extensive regeneration of seedlings; however the rate of regeneration and survival varied considerably among sites. No seedling establishment was observed at equivalent undisturbed sites.

Forest management practices such as clear-felling of forest stands, commercial and non-commercial logging of trees and fuel reduction treatments which substantially disturb the natural ecological balance will have significant effects on ecosystem structure and function (Borg et al. 1988). Hence tree thinning may be considered as a disturbance in the forest, along with logging. Forest thinning is considered a major disturbance factor and has multiple effects on forest ecosystems. Depending on the harvest system used, thinning will generate considerable volumes of residues on site that can kill existing vegetation and also slow down its re-establishment (Metzer and Schultz 1984, Newton and Cole 2006). On the other hand, use of heavy machines during thinning operations and hauling of logs influences both physical and chemical properties of the soil by breaking the hard pan and mixing the soil horizons, creating favourable conditions for seed germination, regeneration and seedling establishment (John et al. 2002). However, the specific responses of the jarrah forest ecosystem to thinning disturbance are not well understood.

After disturbance by logging, there is generally growth of the retained trees and regeneration of both the overstorey and understorey species. The initial effects that logging had on increasing water yield are therefore likely to be transient (Stoneman and Schofield 1989). However the response of forest components varies with forest type and more specifically to the methods of thinning (Peacock 2008). Ground-based machine logging disturbs the soil and vegetation directly (Peacock 2008). However, minimal physical disturbances are expected if the tree felling is carried out by chain-saw method unless the felled trees are dragged out of the forest by heavy machinery. Furthermore, direct physical disturbance is rare if the trees are killed in a standing position by making a notch with an axe and applying a prescribed herbicide to cause tree death. In the latter case, an immediate effect of tree thinning on structure and composition of the forest can be expected in stand density and basal area of the live trees with minimal impact on understorey species richness and ground cover. In this respect the herbicide method of tree killing will have a different form and level of disturbance than felling as a means of tree thinning.

2.1 Thinning effects on stand structure and composition

Thinning is reported to affect the vegetation structure of a forest in various ways. Most of the studies conducted in North American and Canadian softwood forests typically correlate changes in understorey vegetation structure and composition after thinning to the changed light regimes following overstorey thinning (Metzger and Schultz 1984, Alaback and Herman 1988). More recent studies have argued that thinning-induced changes in vegetation structure and composition are simply a response to transient increases in resource levels, and not to the direct thinning-induced vegetation structural changes (Lindh and Muir 2004). However, some other studies have focused on changes in understorey vegetation structure and composition leading to development of a more diverse flora and fauna after thinning and related these to the altered microclimate, soil condition and typically the water and nutrient availability (Hanley 2005, MacCracken 2005). By changing microclimatic condition, thinning also alters important eco-physiological and ecological processes (Aussenac 2000, Thibodeau et al. 2000, Bauhus et al. 2001). In a study conducted in a *Eucalyptus sieberi* regrowth stand in East Gippsland six years after thinning and fertilization, Bauhus *et al*. (2001) demonstrated that the photosynthetically-active radiation above the understorey was 41 % of full sunlight in thinned stands and 32-34 % in unthinned stands. This suggests that the canopy density had not returned to the pre-treatment levels even six years after thinning. However, light attenuation within the understorey did not differ among the treatments, confirming that the understorey cover had not been seriously disturbed during thinning nor increased in response to increased light and nutrient availability.

Some research has demonstrated that thinning increases stem diameters (Stoneman et al. 1996, Brissette et al. 1999, Pothier 2002, Sullivan et al. 2002), increases crown size (Brissette et al. 1999, Lindgren and Sullivan 2001) and decreases mortality of the retained trees (Brissette et al. 1999). Stoneman et al. (1996) reported that thinning and fertilizing increased the growth rate of pole-sized jarrah trees and stands. This study was conducted in the dry eucalypt forests of south-west Western Australia at Inglehope Forest Block, near Dwellingup, classified as a high rainfall zone within dry eucalypt forest. Five thinning treatments were examined with retained basal area 5.5, 10.9, 16.4, 22.4 and 28.5 m^2 ha^{-1}, respectively and two fertilizer treatments (unfertilized and fertilized with 400 kg ha^{-1} nitrogen plus 229 kg ha^{-1} phosphorus) to examine effects on growth and water relations of pole-sized jarrah trees. Results indicated that diameter growth rate of jarrah trees increased with decreasing stand density. Growth rate was greatest with the lowest stand density but the increase was restricted to fastest-growing 200 stems per hectare. In addition, stand growth efficiency i.e. growth per unit leaf area index (LAI) increased in response to thinning. This result was biased in the sense that the slower growing trees had been removed from the heavily thinned plots; whereas they remained in the unthinned and lightly thinned plots and hence the average growth rate was lower for unthinned and lightly thinned plots.

2.2 Understorey vegetation response to thinning

Understorey response to thinning is likely to depend on several factors such as pre-treatment site condition, types of site disturbance, season and intensity of thinning and the extent to which the overstorey structure is directly modified (Abella and Covington 2004). It is observed in some research that thinning has increased species richness (Wienk et al. 2004, Metlen and Fiedler 2006) while others have recorded reduced species richness in dry conifer forests (Metlen et al. 2004). The increase in species richness after active restoration treatments in ponderosa pine/ Douglas-fir forests in western Montana forest is considered to be due to the ability of the shrub species to respond to increased light availability and the reduced underground competition from the overstorey (Metlen and Fiedler 2006). Research evaluated four treatments i.e. no action (control), silvicultural cutting (thin-only), spring burning (burn-only), and silvicultural cutting followed by spring burning (thin-burn) on the understory community (Metlen and Fiedler 2006). However, reduction in species richness and cover was likely due to physical disturbances to the growth resulting from fallen debris comprising tree canopies and branches generated by the thinning operations (Metlen et al. 2004).

Studies of overstorey thinning effects on understorey vegetation have produced mixed results. It is observed that there is a wide divergence in opinion from northern hemisphere studies on the effects of thinning on understorey vegetation. Contrasting examples of responses include: minimal effects on understorey vegetation (Sullivan et al. 2002, Sarkkola et al. 2005); the effect is short-lived and transient in nature (Thysell and Carey 1999); the effects can be noticed only in distinct pulses categorized as intermediate, short-term and long-term gradual (Moore et al. 2006). Still others have argued that the longer-term pattern of understorey succession is not altered by thinning (Metzer and Schultz 1984, Alaback and Herman 1988). At the landscape scale, thinning will lead to a greater heterogeneity in the spatial structure of the forest stands, but within those stands, will lead to a more homogenous structure (Montes et al. 2005). Other studies consider that thinning will alter the relationship between overstorey species (Ball and Walker 1997), stimulate understorey growth and development (Tappeiner and Zasada 1993, Albrecht and McCarthy 2006) and alter species' relative covers (Ward 1992, Zenner et al. 2006). Species which responded to thinning in one study (Falkengren-Grerup and Tyler 1991) have been reported to be unresponsive in another study (Graae and Heskjaer 1997).

Most of the above studies were conducted in temperate forests of the northern hemisphere where light and temperature were the most limiting resources (Metlen and Fiedler 2006). Metlen and Fiedler (2006) observed that treatments differentially impacted the understorey community of ponderosa pine/Douglas-fir forests in Western Montana, with the most dramatic changes in the thin-burn treatment. They evaluated the effects of no action (control), silvicultural cutting (thin-only), spring burning (burn-only), and silvicultural cutting followed by spring burning (thin-burn) at both plot (1000 m^2) and quadrat (1 m^2) level. Data were collected before treatment and three subsequent years after treatment. The burn-only treatment initially reduced richness and cover of the understorey, but by year three all active treatments increased plot-scale understorey richness relative to pre-treatment and control. Simpson's evenness increased for the first growing season after burning, but was not influenced by treatments in subsequent years. Forbs, both native and exotic, were the most responsive life form and increased in richness and cover after thinning, with the greatest response in the thin-burn treatment.

The impacts of logging and prescribed burning on jarrah forest properties, including short-term impacts of logging on understorey vegetation in a jarrah forest, were studied by Burrows et al. (2001) while the effects of thinning and burning operations in 10 to 13-year-old rehabilitated bauxite mines in the jarrah forest were reported (Grant and Norman 2006). As part of an investigation into the ecological impacts of two silvicultural treatments, gap cutting and shelterwood cutting (Burrows et al. 2001), a survey was conducted 4 years after logging to examine the short-term effects of these logging treatments on understorey species richness and abundance in the jarrah forest located about 25 km north-east of Manjimup in the south-west of Western Australia. Native plant species richness in unlogged coupe buffers was similar to that in the adjacent logged patches. However, the mean number of species per m^2 was 20-30 % higher in the unlogged buffers than the logged patches. At all sampling scales the abundance of native plants was 20-35 % higher in buffers, while the abundance of introduced (weed) species was significantly higher in the logged patches.

A study was conducted by Grant and Norman (2006) in 10 to 13 year-old rehabilitated bauxite mines in the jarrah forest to investigate the effects of thinning and burning operations. Four sites at Jarrahdale and Huntly mines were subjected to a complete factorial treatment of two burning (autumn and no burn), four thinning (control, light, moderate and heavy) and two fertiliser (none and 500 kg ha^{-1} DAP) treatments. Thinning treatments equated to retention of 1,750 stems ha^{-1}, 1,111 stems ha^{-1}, 625 stems ha^{-1} and 400 stems ha^{-1} for the respective thinning intensities. Thinning operations occurred in January 2002, followed by autumn burning in May 2002. Burnt sites were fertilised in August 2002 and unburnt sites in August 2003. Pre-treatment and 18 months post-treatment measurements were taken within 20 x 20 m plots for tree height, diameter, bark thickness, bole length, form and health; understorey composition, density and cover; fuel loads; vegetation structure; soil nutrients; groundwater level; and leaf area index. Both thinning and burning successfully removed the prominent mid-storey component of the vegetation structure, resulting in a two-tiered structure characteristic of the native jarrah forest. Across all treatments, the majority (83 %) of the species present in the pre-treatment vegetation were present 18-months post treatment. The average Sorenson similarity to the pre-treatment vegetation was 56 % in burnt sites and 62 % in unburnt sites. In unburnt sites, similarity to the pre-treatment vegetation significantly reduced from 75 % in the unthinned treatment to 56 % in the heavy thinning treatment. Similar reductions occurred in burnt sites. Thinning had no other effect on the understorey, while burning significantly affected over half of the measured understorey vegetation characteristics. Burning increased native and weed species richness, but reduced native species diversity and evenness. Burning also increased the density of various fire-response groups, notably the auto-regenerating long-lived resprouters. Fertiliser application increased the density and cover of weeds.

2.3 Extent, structure and composition of dead woody debris in relation to thinning

Dead woody material, referred to as dead woody debris (DWD; also known as coarse woody debris) in this study, is an important structural component of many forest ecosystems, and has a significant role in ecosystem functioning (Harmon et al. 1986). Dead WD may include whole fallen trees, fallen branches, logging residues such as fragmented logs and branches, stumps and standing dead trees called snags. Several studies conducted in American and Canadian softwood forests have reported that DWD plays an important role in carbon sequestration, nutrient cycling, soil stability, hydrology, soil forming processes and soil retention capacities (Harmon et al. 1986), and wildlife habitat (Hagan and Grove 1999). In addition, dead woody material offers rooting substrate for plants (Harmon and Franklin 1989) and plays a vital role in nutrient cycling and carbon storage (Brown et al. 1996, O'Connell 1997, Berg 2000). Some recent Australian research publications have discussed DWD as wildlife habitat (Meggs 1996, Lindenmayer et al. 1999, Grove and Meggs 2003), described attributes and amounts of DWD (Lindenmayer et al. 1999, Grove 2001) and the decomposition of DWD (Brown et al. 1996, O'Connell 1997) in Australian forest management systems. It is reported that the quantity of DWD in managed forest is between 2 and 30 % of the quantity in unmanaged forestland in Sweden (Fridman and Walheim 2000).

There are relatively few studies with special focus on DWD in Australian forest ecosystems. Some of them have concentrated their effort on amount, structure and composition of DWD materials and dynamics of DWD in old-growth forests (Lindenmayer et al. 1999, Grove 2001). A review of published and unpublished literature up to the year 2002 on DWD quantities is presented by Woldendorp and Keenan (2005). This review of previous literature has provided a basis for further studies on the DWD attributes and its importance in the study of carbon and nutrient stocks, forest dynamics; habitat values and the assessment of fire hazards. Very few of these previous studies have reported the structural differences of the DWD in relation to the method and intensity of thinning; also lacking is consistency among studies in the classification of DWD resources relative to both standing and lying dead wood components of the forests (Woldendorp and Keenan 2005).

Quantity and structural integrity of the DWD is highly dependent on the type of the forest, size of the individual trees and the number of stems per unit area. It is also dependent on the durability of the wood, disturbance regimes, decomposer activity and climatic conditions that determine decomposition. Generally, it is expected that re-growth forest with small-statured trees contain lower quantities of DWD than old-growth large-statured forests. Quantity and structural composition of the DWD in the forests can differ largely depending up on the silvicultural practices applied i.e., whether the thinning residues are left or completely removed from the site. A review of DWD in Australian forest ecosystems reveals that the tree species in Australian forests have wood debris with higher density and durability than softwoods or hardwood species in other temperate regions (Woldendorp and Keenan 2005).

2.4 Conclusion

The review of the pertinent literature from Australia and elsewhere related to thinning effects on stand structure and composition, understorey vegetation, and DWD provides context for this study of thinning effects on forest structure and composition in the Wungong Catchment, Western Australia. Specifically, it provides the basis to better understand and interpret the results of this study within the broader body of knowledge on this topic in Australia and elsewhere. The presentation that follows is structured such that Chapter 3 investigates the short-term effects of tree thinning on stand structure and composition especially the species composition, stand density and basal area for three overstorey trees species, understorey species richness and ground cover in relation to the thinning intensity. Chapter 4 investigates how thinning affects the amount, structure and composition of DWD components of the forest. Finally in Chapter 5, findings of the research are synthesized and discussed.

Chapter 3: Thinning effects on stand structure and composition

3.1 Introduction

In Australian forest ecosystems, several studies have documented natural as well as human-induced disturbances that caused significant changes in post-disturbance forest properties (Pook et al. 1966, Abbott 1984, Ruprecht et al. 1991, Stoneman 1993, Stoneman et al. 1996, Brown 1997). Previous authors have described the impacts of natural disturbances such as occurrence of fire events (Morrison et al. 1995, Bradstock et al. 1997, Fisher et al. 2009); insect infestation (Campbell 1962); disease outbreak (Podger 1972, Batini 1973, 1974); extended drought (Pook et al. 1966, Hnatiuk and Hopkins 1980); and flood and severe storms (Unwin et al. 1988, Bren 1993) on different components of forest ecosystems while others have described forest management activities such as fuel reduction treatment and forest thinning treatments that have significant effects on post-treatment vegetation properties (Christensen and Kimber 1975, Borg et al. 1988).

Several studies have reported about the post-disturbance ecology of the Australian forest ecosystem (Ashton 1979, Beard 1990, Kealley 1991). Large areas of *Eucalyptus salmonophloia* woodlands around the Coolgardie and Kalgoorlie mining centres are now regrowth woodlands following clearfelling to supply mining timber and fuelwood (Kealley 1991). Clearing of trees began in 1890 and continued until 1960 (Kealley 1991). Significant seedling recruitment occurred following severe disturbance with seed arising from the remaining trees and canopy residues left after logging (Kealley 1991). However, regeneration was not always successful and some regrowth woodlands are very sparsely vegetated in the areas following disturbance (Beard 1990).

In many Eucalyptus spp. which recruit seedlings after fire, studies of seed dynamics in inter-fire periods have shown that firstly, several seasons of seed are stored in woody fruits on the plant forming a substantial seed reserve; secondly, there is a constant seed fall from the canopy and; thirdly that the seed predators such as ants eat most of this seed and prevent the establishment of a soil seed bank (Ashton 1979, O'Dowd and Gill 1984, Anderson and Ashton 1985, Anderson 1988). Evidently fire interacts with these life history attributes to provide an environment suitable for seedling recruitment in three ways. Firstly, fire causes a mass release of all canopy-stored seeds, satiating predators and thereby facilitating the establishment of a soil seed bank (Ashton 1979, O'Dowd and Gill 1984, Anderson 1988). Secondly, changes in soil condition following a fire can increase rates of germination and establishment (Burrows et al. 1990). This has been attributed to the addition of nutrients from ash and elimination of plant pathogens in the soil (Loneragan and Loneragan 1964). Finally, there was an increase in seedling survivorship associated with an increase in resource availability particularly in gaps created by the death of adult plants (Wellington 1981, Wellington and Noble 1985b). Perhaps this model of post-disturbance ecology applies to the present study of effects of prescribed silvicultural treatments comprising thinning of the forest stands by selective removal of trees by felling and/ or herbicide treatment of trees to kill them. Significant changes in the overstorey stocking density with minimal physical disturbance to soil properties are expected to significantly influence the resource levels available for the remaining vegetation, thereby affecting the post-thinning ecological processes.

Thinning may have multiple effects on stand structure and composition of the forest (Borg et al. 1988). For example, Borg et al. (1988) conducted a study in the forest of Western Australia from 1976-1985 that investigated the effects of heavy selection cutting and clear-felling followed by regeneration on groundwater, streamflow and stream salinity in four small catchments (Crowea, Poole, Iffley and Mooralup). The catchments were logged between November 1976 and March 1978. Hydrological investigations conducted for eighteen months after completion of the logging operation indicated that regeneration had already begun. During this study period, the annual rainfall in the region was generally below the long-term mean. This probably influenced the magnitude and duration of the hydrologic response to logging and regeneration, but not the general trends. Ground water levels rose for two to four years after treatment and then started to fall again. This transient increase in water level was expected to be followed by a return to pre-treatment values within 15 years after the beginning of regeneration. Similarly stream flow also increased for two years and then gradually declined as the vegetation regenerated. Flow-weighted mean annual stream salinities rose for one to three years after treatment but have declined subsequently.

In the jarrah forest, there has been considerable work on the effects of forest thinning and logging on different aspects of forest ecosystems such as growth and water relations of jarrah stands (Stoneman et al. 1996), hydrological response to thinning in a small jarrah forest catchment (Ruprecht et al. 1991, Ruprecht and Stoneman 1993, Stoneman 1993) and survival of jarrah and marri habitat trees after logging (Whitford and Williams 2001). Thinning trials have demonstrated the capacity to dramatically increase both water and timber yields (Stoneman and Schofield 1989, Stoneman et al. 1996, Water and Rivers Commission 1997, Bari and Ruprecht 2003). However, the water output increased by thinning may return to the pre-thinning levels within 14 years after treatment if periodic follow-up treatment of coppice is not carried out (Bari and Ruprecht 2003). Similarly, both stand growth efficiency and average tree growth efficiency (growth efficiency is the amount of stemwood produced per unit of leaf area) showed large increases in response to thinning and fertilization in an even-aged regrowth *E. marginata* stand in the northern jarrah forests of Western Australia (Stoneman and Whitford 1995). The potential to encourage fewer but larger trees and a more open canopy after thinning may lead to slight changes in the local numbers of some herbs, ground cover and understorey species richness (Stoneman and Whitford 1995, Mattiske Consulting Pty Ltd. 2005). These slight changes in species richness and ground covers are well within the range of natural fluctuations that occur on a local scale.

In a study conducted at the Inglehope Forest block with differential basal area reduction and fertilization treatments (T1= 5.5, T2 =11, T3 =16.5, T4 = 22.5 and T5 = 28.5 m^2 ha^{-1} basal area under bark) × two fertilizer treatments (F0 = unfertilized and F1= fertilized with 400 kg ha^{-1} Nitrogen plus 229 kg ha^{-1} Phosphorus), Stoneman et al. (1996) demonstrated that diameter growth rate of *E. marginata* trees increased with decreasing stand density (0.02 cm $year^{-1}$ in highest stand density treatment and 0.75 cm $year^{-1}$ in the lowest stand density treatment); however, the increase was limited to the fastest-growing 200 stems per hectare in the stand. Application of fertilizer increased average diameter growth in all thinning treatments, with the greatest increase in rate (1.2 cm $year^{-1}$) occurring in trees growing at the lowest stand density. Thinning reduced leaf area index to 2.1 in the T4 and T5 treatments to 0.8 in the T1F0 treatment, resulting in less water stress and increased the stand growth efficiency. This study was conducted in a high rainfall area with an average annual rainfall of 1,217 mm at Dwellingup.

Many Australian thinning studies have focussed on the response of the overstorey (Connell and Raison 1996, Brown 1997), an approach which focuses on retained trees and not the component lost after thinning treatment (Reader 1988). Comprehensive studies on the effects of prescribed thinning treatments on stand structure and tree composition of jarrah forest of Western Australia were rarely reported. However, a comparative study conducted between virgin and cut-over jarrah forest in Western Australia is relevant to the issue of thinning effects on forest structure (Abbott 1984). Moreover, ecological impacts of thinning on forest re-growth are relatively poorly known as are the complexities of multi-species interactions and the responses in the understory (Bailey and Tappeiner 1998).

Reduced rainfall in the past decades in south-west Australia, leading to a decline in the ground water table level and also reduced levels of water in most of the drinking water reservoirs, has forced water management authorities to look for new sources of water supply to fulfil the increasing water demand of the Metropolitan area (Water Corporation 2005). Similarly increasing demands for forest resources such as timber and fuel, combined with a desire to maintain ecological sustainability of forests has led to interest in a greater diversity of silvicultural practices, including both "ecologically friendly" practices and intensive forest management (Tappeiner et al. 1997). The effectiveness of forest thinning practices for meeting water production along with ecological objectives has become of great importance to water management bodies, forest scientists and managers.

More recently, some areas of the Wungong catchment comprising a high proportion of jarrah forest have been subjected to adaptive forest thinning trials followed by prescribed burning with a view to increase both water and environmental benefits (Water Corporation 2005). Notwithstanding the interest in managing forests for water output, effects of forest thinning on post-thinning structure and composition of the forests are poorly known and cannot be neglected. Hence, forest thinning practices should be implemented in conjunction with the sustainability principle of forest management so that there will be acceptable impacts on forest stock, function and ecosystem health in the long-term.

The aim of this chapter was to assess the effects of tree thinning on ecosystem structure of the jarrah forest of the Wungong catchment of south-west Western Australia. More specifically, this study focussed on the effects of two different thinning strategies on stand structure and composition of the jarrah forest. Thinning in this study is defined as the silvicultural treatments that kill selected trees to improve the growth of retained trees and allow the catchment to produce extra run-off. Two types of forest thinning strategies were implemented i.e., commercial thinning followed by herbicide treatment of selected trees to kill them (log+notch) and herbicide treatment (notch-only). Commercial thinning in this trial is a method of cutting down and trimming trees and removal of the millable bole from the logging sites to produce commercial wood products whereas non-commercial thinning included killing of trees in standing position by stem injection with glyphosate herbicide.

3.2 Materials, methods and study area

3.2.1 Research site location and description

The Wungong Catchment, which is mainly State forest, lies on the western margin of the laterite-capped granite and gneiss rocks of the Darling Plateau (Churchward and McArthur 1980), located in the south-western fringes of the mostly arid and ancient Great Plateau (Dell et al. 1989). The Wungong catchment lies on the deeply weathered landscapes of the northern parts of the Darling plateau (Figure 1). It drains in to the Wungong Brook, which developed following weathering of the uplifted Plateau. The Wungong Catchment has a total area of 12,845 hectares.

The dominant soils of the Wungong forest are derived from various elements of the laterite profile and comprise highly weathered materials depleted of many nutrients (Turton et al. 1962, Gilkes et al. 1973). The soils of the Wungong catchment area are mostly lateritic gravels consisting of up to 5 metres or more of ironstone gravels in a yellow sandy matrix. Related soils are lateritic podzolic soils with ironstone gravels in a sandy surface horizon overlying mottled yellow-brown clay subsoil. These materials frequently overlie a pallid zone up to 30 metres or more in thickness (Beard 1990). However, the surface soil is darkened by the addition of organic matter that supports early growth and development of seedlings.

The Wungong catchment is characterised by a dry Mediterranean climate with 1000-1400 mm annual rainfall, falling mostly in winter while almost half of the year experiences dry and

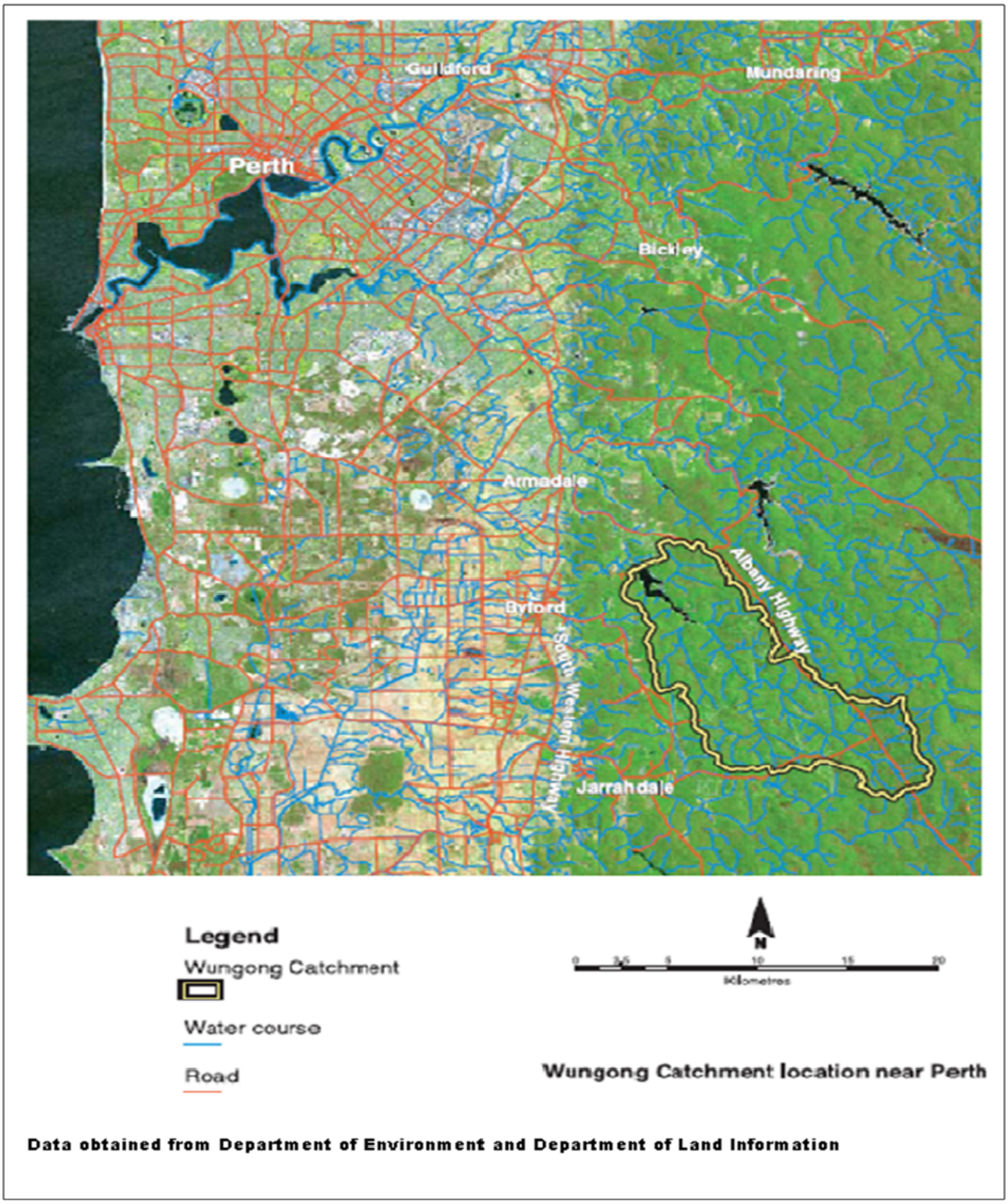


Figure 1 Location of Wungong Catchment near Perth, Western Australia.

hot conditions (Beard 1990). The annual rainfall for the Jarrahdale weather station (averaged from 1997-2003), which is representative of the Wungong catchment, was 985 mm. Rainfall has decreased at Jarrahdale with a 15 % decline since 1975 and 8 % reduction after 1996 (Mattiske Consulting Pty Ltd. 2005). More recent data obtained from the Jarrahdale weather station suggests that there was an increase in annual rainfall to 1133.8 in 2009 with a sharp reduction in 2010 rainfall to about 614.8 mm, the lowest in the last three decades.

Much of south-western Australia supports evergreen sclerophyllous forests 20-30 metres tall, dominated by jarrah and marri (*Corymbia calophylla*) across a range of climatic and edaphic gradients (Havel 1975a, Wardell-Johnson and Horwitz 1996, Wardell-Johnson et al. 1997). On the ridges throughout the forest four other small tree species occur widely: *Banksia grandis* Willd; *Allocasuarina fraseriana* (Miq.) L. Johnson; *Persoonia longifolia* R. Br. and *Persoonia elliptica* R.Br. Only two eucalypts, jarrah and marri form the upper storey in most of the forest. Depending on site quality, the top height of the jarrah forest overstorey varies from 20 to 30 m and canopy cover varies from 25-50 %. The sclerophyllous understorey comprises a rich diversity of woody shrubs and perennials as well as annual herbs ranging from 0.5 to 2 m in height, with understorey cover extending from 25 to 80 %. The undergrowth is predominantly from the families Anthericaceae, Dasypogonaceae, Leguminaceae, Orchidaceae, Apiaceae, Epacridaceae, Asteraceae, Proteaceae, Restionaceae, Myrtaceae, and Cyperaceae.

The vegetation of the Wungong catchment is characterized by a mixture of jarrah and marri in most of the dryer areas. However, in wetter areas this association is replaced by one containing the blackbutt (*Eucalyptus patens*), bullich (*Eucalyptus megacarpa*), paperbark (*Melaleuca quinquenervia)* and swamp banksia (*Banksia littoralis*) (Heddle et al. 1980). The Wungong forest is a dry sclerophyll open type re-growth forest developed after periodic logging and clear-felling of the forest stands. This resulted in a very dense regrowth forest dominated by jarrah, marri and banksia on the ridge top and slopes and a mixture of yarri and flooded gums in the main valley floors (Environmental Protection Authority 2005).

3.2.2 Previous catchment management environment

The West Australian forest remained unlogged until the 1830's when the Europeans settled the state. However, commercial harvesting of jarrah forest intensified after 1870 and has continued to the present. Data obtained from the Department of Conservation and Environment (DEC, previously known as Conservation and Land Management, CALM) indicated that most of the Wungong catchment was first logged before 1929, with few, if any controls, as the Forest Department was not established until 1919 (Terry et al. 2005). CALM reports show that large areas within the Wungong Catchment were logged under tree-marking during 1930-49 (10,600 ha) and 1970-89 (7,200 ha). Overall 9 % of the catchment area has been logged once, 26 % twice, 57 % three times and 6 % four times (Water Corporation 2005).

Thinning of the forest has been carried out for many years by foresters to improve crop tree growth. Potential trees suitable for producing habitat hollows would reach a suitable size earlier if selected surrounding trees were thinned. However, uncontrolled thinning was not conducted within the drinking water catchments because of concerns about water quality and quantity raised first in the 1920s and then again from the 1970s to the present (Lee and Abbott 2004).

Forest management planning in the 1980s used a wide range of variables, such as rainfall and landform, to demarcate zones for priority uses. This approach assigned a priority use to each area and described all forest activities as either compatible or incompatible for that zone (Forest Department of Western Australia 1980). For example, the higher rainfall zones had a water production priority with forestry as being compatible, but regeneration of dense forest or water polluting land-use was incompatible. In the 1990s, forest management shifted to multiple-use planning with no obvious priority for land-use. As in the past, the actual priority was timber production since this was the core business of those responsible for implementing the plans. The current Forest Management Plan 2004-2013 continues the multiple use planning approach, now with an emphasis on conservation and biodiversity (Conservation Commission 2004).

The Wungong catchment thinning trial funded by the Water Corporation commenced in 2005 with a view to enhance water availability, biodiversity and timber values in compliance with the Forest Management Plan 2004-2013. To achieve these objectives commercial thinning, non-commercial thinning of cull trees, control of coppice and excessive regrowth and regular prescribed burning were proposed.

3.2.3 Research design and thinning intensity

The jarrah forest selected had been logged well before 1960 in the Vardi Road sub-catchment. An experimental block (5.7 ha) was established in the Vardi Road sub-catchment (Figure 2) using a Latin square design (9 individual plots of 90 x 70 m each), with triplicate plots for the two thinning methods and for the control (untreated) forest (Figure 3). Treatments were assigned randomly; however, plots proposed for logging treatment were arranged diagonally from plot TA1 to TA3 to minimize physical disturbance to other treatment plots during removal of logs classed as suitable for timber. Before the commencement of thinning treatments, tree markings were carried out in February 2009 by DEC staff following the interim guideline for silvicultural practice in the jarrah forest of the Wungong Catchment (Department of Environment and Conservation 2007). Furthermore, on-ground operations within the Wungong Catchment, including notching and felling of the trees with non-commercial value were carried out by DEC. Forest Products Commission undertook all commercial harvesting operations within the catchment as part of the normal northern jarrah forest harvesting program.

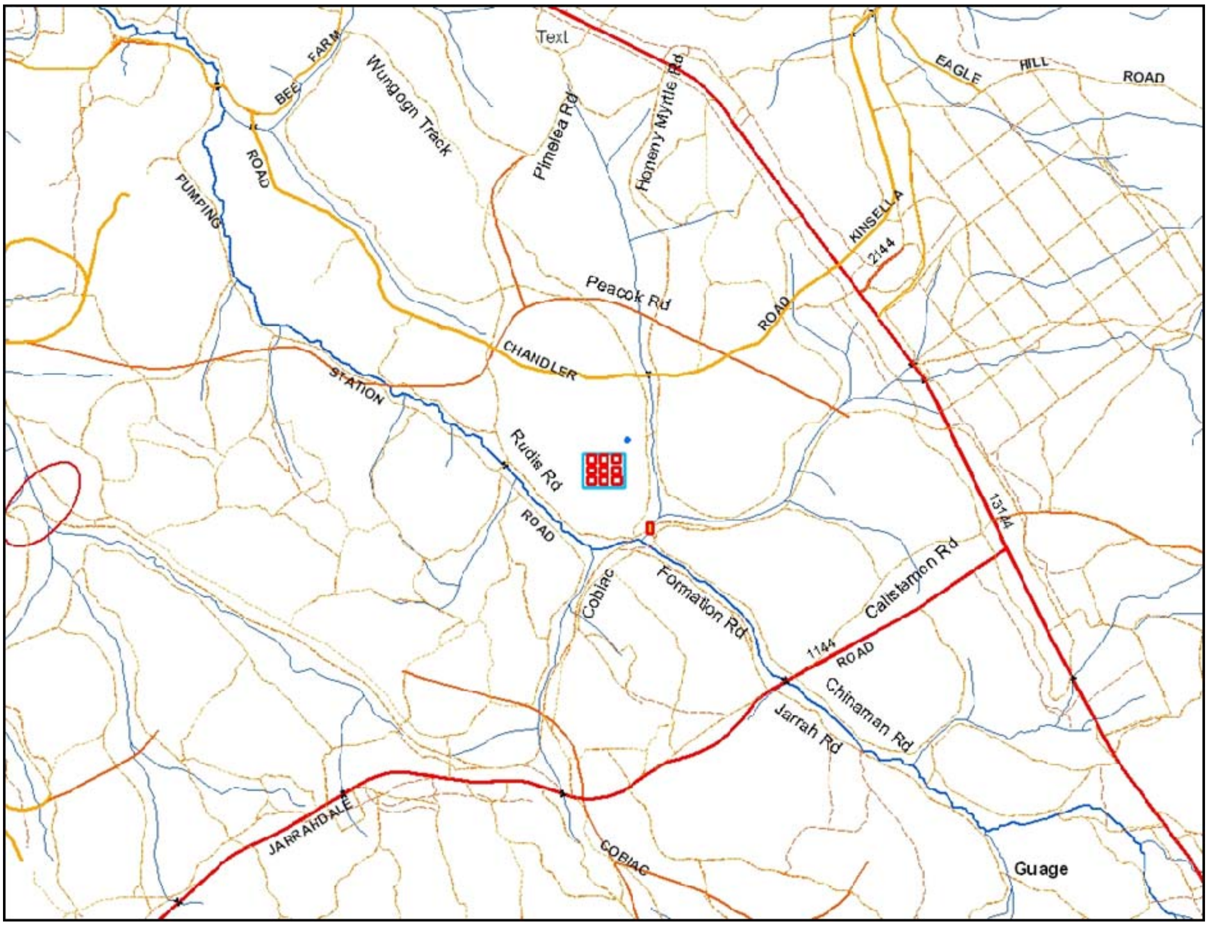


Figure 2 Site location of the research plot in the Vardi Road sub-catchment, Western Australia.

In general tree marking is the means by which stand objectives are marked out in the forest so that harvesting and treatment operations can proceed. By marking trees to be retained, a vision of the future development of the stand is provided. Before marking commences, the first task in marking a patch of trees was to determine the appropriate silvicultural treatment (thinning, regeneration release, shelterwood or 'selective cut in dieback') and whether patch boundaries were apparent. The process for making these decisions is outlined in "Treemarking and Silviculture in the Jarrah Forest" (Bradshaw 1987).

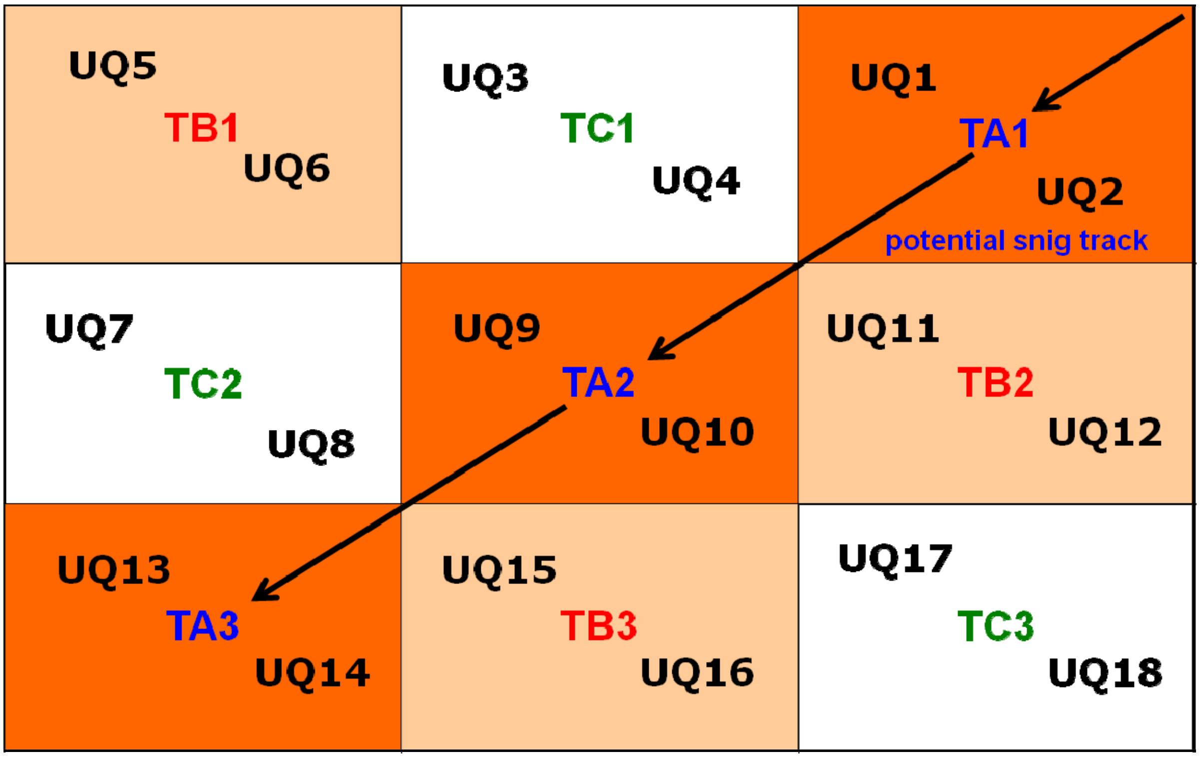


Figure 3 Study plot design with planned snig track for log removal (solid line). UQ = understorey study quadrats (2×2 m), TA = log+notch treatment (orange colour), TB = notch-only treatment (yellow colour) and TC = control treatment (white colour) (90×70 m).

Tree markings were carried out with an objective to reduce the total basal area to 15-18 m^2 ha^{-1} as prescribed in the DEC guidelines (Department of Environment and Conservation 2007) (Figure 4). To achieve this target basal area, only small trees, diseased and deformed trees and some selected trees classed as suitable for timber were marked for culling (Cull), while potential future timber trees were marked for retention (Crop). Similarly, large trees either already having hollows or the potential to produce hollows were marked as habitat (Habitat). Some trees which were selected for long-term monitoring were marked for retention as research trees (Research) during the pre-thinning forest inventory. Some plant species such as balga (*Xanthorrhoea preissii*) and bull banksia were left untreated as prescribed in the silvicultural guidelines for biodiversity conservation (Department of Environment and Conservation 2007).

Figure 4 Trees were marked before thinning of the research plots. Trees marked with yellow stripe on trunk were proposed for logging (a), trees marked with white stripe as potential crop tree (b), trees marked with letter <u>H</u> as habitat tree (c) and all the trees without any marking were proposed for notching (d).

Basal area (BA), in this thesis is the sum of the cross-sectional areas of trees greater or equal to 10 cm in diameter outside bark for a particular plot area measured at 1.37 m above ground level and is usually expressed in m^{-2} ha^{-1}. Basal areas for individual trees were calculated by using simple arithmetic formula BA = $\pi d^2/4$, where BA = basal area, π = 3.141592654, and d = diameter at breast height (DBH) of individual trees.

Two thinning treatments were imposed i.e., selective felling for timber followed by stem injection of herbicide (log+notch) or stem-injection of herbicide only (notch-only) (Figure 5). Trees were felled using a chain-saw for the selected trees suitable for logging. The rest of the cull trees were notched with a small axe and glyphosate herbicide was injected in the notch. The concentration of the glyphosate used was 360 g/ litre and was used directly without dilution at a dose of 2-3 ml/ notch (cut a notch on the trunk and inject glyphosate), at every 10-15 cm around the stem. Prescribed thinning treatments were applied in September 2009, for a planned basal area reduction of 70.7 % and 53.6 % for log+notch and notch-only treatments, respectively (Table 1). Actual thinning intensity received by log+notch and notch-only treatment plots were 64.4 % and 49.4 %, respectively (Table 2). Mean log volume of 37.5 m^3 ha^{-1} (41.1, 40.6 and 30.9 m^3 ha^{-1} from plot 1, 5 and 7, respectively) was removed from the log+notch treatment leaving the logging residue (i.e. the crown of the tree) on site where it fell.

3.2.4 Sampling methods

3.2.4.1 Tree census in the study plots

Corner locations of the research block were marked with a hand-held geographical positioning system (GPSMAP 60CSX, GARMIN, USA). The study site was located between 116° 10′ 19.27″ - 116° 10′ 29 59″E and 32° 17′ 54.60″ - 32° 17′ 54.60″S). To avoid edge effects, measurements were taken within the central 50×40 m area leaving 20 and 15 metre buffers along the margins of each plot.

Measurements were taken before and after the implementation of thinning treatments. Pre-thinning baseline forest conditions assessed were tree number and size for each species, tree height and crown base height which were measured in March 2009 in order to assess response to thinning in the post-thinning forest system. To capture thinning effects on the above parameters, one year post- thinning measurements were carried out in April 2010. Within each plot (50×40 m area), diameters were measured and species identified for all trees with ≥ 10 cm diameter at DBH. Trunk diameters were measured at 1.37 m above the ground level. Additionally, tree height for 168 selected jarrah and marri trees representing upper, mid and understorey canopy were measured using a digital hypsometer (FORESTOR VERTEX, Forestor AB Sweden).

Figure 5 Details of thinning methods- logging by chain-saw (left) and stem injection of glyphosate by making notch in the trunk (right).

Table 1 Basal area (BA, m² ha^{-1}) measured for trees marked as Cull, Crop, Habitat, and Research trees in each plot of the Murdoch Research block at Vardi Road sub-catchment. The planned BA reduction is indicated by the difference between total BA (All trees) and BA of all retained trees. NA = not applicable. See also Table 2 for actual BA reduction.

Plot	Treatment	Cull	Crop	Habitat	Research	All trees	All retained trees
1	Log+notch	36.7	8.0	5.3	1.3	51.2	14.6
2	Control	NA	NA	NA	0.4	35.6	35.6
3	Notch-only	24.4	16	5.2	2.9	48.5	24.1
4	Control	NA	NA	NA	0.4	35.8	35.8
5	Log+notch	34.9	4.6	7.1	2.6	49.2	14.3
6	Notch-only	22.0	12.7	1.5	1.8	38.1	16.1
7	Log+notch	34.8	4.5	10.1	0.7	50.1	15.3
8	Notch-only	23.9	15.3	4.3	1.2	44.8	20.9
9	Control	NA	NA	NA	0.2	37.9	37.9
Mean for Log+notch		35.5	5.7	7.5	1.5	50.2	14.7
Mean for Control		NA	NA	NA	0.3	36.4	36.4
Mean for Notch-only		23.4	14.7	3.7	2.0	43.8	20.4

3.2.4.2 Understory vegetation sampling

Understory sampling was carried out in quadrats of 2×2 m area sited randomly within the larger plots to represent an average condition of the site apart from areas directly disturbed by logging in the log+notch plots (Fig. 2). Two permanent sampling quadrats in each plot (50 × 40 m) were established. All the plants including herb, shrub and tree species (< 1 m in height) were recorded and identified. Numbers of individual plants within each species were recorded to identify the most abundant understory species for all quadrats measured. Understory ground cover was assessed visually using cover-abundance scale value modified from Braun-Blanquet (Braun-Blanquet 1932, Poore 1955). Additionally, total ground cover was estimated for all the quadrats with the cover value calculated from individual plant cover measurements. Pre-thinning measurement commenced in late August 2009 just before thinning intervention and 1 year of post-thinning measurements were carried out at three-month intervals.

Table 2 Actual thinning intensity (in m^2 ha^{-1} and as a % of pre-thin BA) measured for the trees after implementation of thinning treatment in each plot of the Murdoch Research block at Vardi Road sub-catchment. The actual BA reduction is indicated by the difference between total BA and BA of all retained trees. NA = not applicable.

Plot	Treatment	Pre-thin BA	Planned BA retained	Actual BA retained	Planned intensity (%)	Actual intensity (%)
1	Log+notch	51.2	14.6	18.8	71.5	63.3
2	Control	35.5	NA	NA	NA	NA
3	Notch only	48.5	24.1	24.1	50.3	50.2
4	Control	37.9	NA	NA	NA	NA
5	Log+notch	49.2	14.3	12.3	70.9	75.0
6	Notch-only	38.1	16.1	17.7	57.7	53.4
7	Log+notch	50.1	15.3	22.5	69.5	55.1
8	Notch-only	44.8	20.9	24.7	53.3	44.8
9	Control	37.9	NA	NA	NA	NA
Mean for Log+notch		50.2	14.7	17.9	70.7	64.4
Mean for Control		36.4	36.4	36.4	NA	NA
Mean for Notch-only		43.8	20.4	22.2	53.6	49.4

3.2.5 Data analysis

3.2.5.1 Tree species composition

Tree species composition was determined for both the pre- and post-thinning treatment plots from the enumeration of individual trees with DBH ≥ 10 cm of all species recorded during the plot inventory. Density is the number of individuals per hectare, and the basal area is the sum of the cross-sectional areas of all individual trees in the plots with DBH $\geq$ 10 cm measured at 1.37 metres above ground level.

3.2.5.2 Forest structure

Forest structure was determined by constructing histograms for each treatment group showing the frequency of occurrence of individual trees in each diameter size class. These were based on the samplings of three plots of 50×40 m for each treatment. Additionally, a logarithmic equation showing the relationship between DBH and tree height was developed. Heights for all the remaining trees with DBH $\geq$ 10 cm were enumerated using the logarithmic equation $Y = 12.243\ln(x) - 20.965$ where, Y = height of the tree in metres, ln = natural log, and X = DBH in centimetres (cm).

The forest stand was stratified into three distinct layers. Hence, all the trees in the study plots were assigned to one of three distinct height categories. All the trees with DBH $\geq$ 10 cm and the height <10 metres were considered as understory, 10- 20 metre as mid-story and > 20 metres as upper story. Mean height for each stratum was calculated as an average from the heights of individual trees enumerated for each stratum.

Understory plant species recorded were allocated to one of the following guilds:

1. Herbs: Herbs are plants with soft, non-woody stems. They have primary vegetative parts and need support of others to grow upright. Herbs may be both annuals and perennials, but no attempts were made to identify species as annual or perennial. Examples include: *Clematis microphylla, Opercularia hispidula* and *Scaevola pilosa*.
2. Shrubs: Shrubs have no main trunk. Branches arise from the ground level. They are woody and have secondary tissue. Shrubs are perennials and usually smaller than trees.

Examples of shrubs include *Acacia urophylla, Hovea chorizemifolia* and *Trymalium floribundun*.

3. Trees: Trees are large plants characterized by one main trunk. They branch on the upper part of the plant, are woody, and have secondary tissue. They are taller than shrubs. Examples of trees include E. *marginata, E. calophylla,* and *Banksia grandis.*

As part of the analysis of treatment effects on forest structure (before-and-after treatment) and among treatments (after treatment), Quadratic Mean Diameter (QMD) was calculated and analysed. QMD, which is one of the simple and elegant parameters for evaluating one aspect of forest structure (Curtis and Marshall 2000), was calculated using the formula

$$QMD = \sqrt{\{BA / (K \times T)\}}$$

where:

BA = basal area in m^2/ha;

K = 0.0000785; and T = trees/ha

To generate the species-area curve from the data sampled, area of individual quadrats (4 m^2/ quadrat) was summed quadrat by quadrat and the resulting cumulative species richness calculated for each 4 m^2 increase in area up to 72 m^2 (i.e. from 18 quadrats in total). The species area curve was plotted as the cumulative area of the quadrats sampled over the study sites against the respective cumulative species richness (See Figure 18).

3.2.5.3 Statistical analyses

For the comparison of stand structure, species composition, stem density and other vegetation parameters among treatment plots, data were subjected to analysis of variance (ANOVA) in the statistical package SPSS Statistics 17.0 (SPSS 2008). Although the plots were arranged as a latin square design the ANOVA was conducted as a completely randomised design.

Understorey data were analysed using time series analysis of variance to test the effects of thinning on species richness and ground cover. Pre-thinning data were used as covariates to reduce the confounding effects of site variations on the post-thinning comparisons. Treatment differences were compared using univariate analysis of variance under the general linear model. Type III sums of squares were employed as there was equal number of replications in each treatment. The Bonferroni test was performed for pairwise comparisons amongst the treatments where significant differences were observed at treatment level ($p < 0.05$).

3.3 Results

Many variables showed significant differences in stand structural parameters measured before treatment, indicating a high degree of structural heterogeneity among experimental treatment plots. The focus of this chapter is on the effects of treatments on structural parameters such as species composition, stand density and basal area distribution, understorey species richness and ground cover at treatment level. There were no significant differences in the understorey species richness and ground cover at treatment level. To understand thinning impacts on species richness and understory ground cover, before and after comparison was also made within the treatments at quadrat level.

3.3.1 Forest structure and composition

3.3.1.1 Stand density and basal area composition by tree species

Co-variates for stand density for 20-30, 30-40, 40-50, 50-60 and 60-70 cm and stand basal area for 20-30, 30-40, 40-50 and 60-70 cm diameter classes appeared to be significant ($P < 0.05$) in the model (Table 3). At $P < 0.08$, there were also pre-existing differences in jarrah and marri basal area and mid-storey and upper storey density that may have confounded the results if the co-variates were not included in the analysis.

There was a significant difference ($p < 0.05$) in the mean stand density [F (3, 5) = 12.428, p = 0.009] and basal area [F (3, 5) = 19.574, p = 0.003] among the treatment groups (Table 3). Notwithstanding the pre-thinning differences among treatment plots, both the mean stand density and basal area in control plots were significantly higher than log+notch and notch-only treatments but log+notch and notch-only treatment did not differ from each other (Figures 6 and 7). Mean density and basal area for banksia tree species among the treatment groups were not different as thinning was not targeted to this species. However, there were highly significant differences in the mean stand density [F (3, 5) = 21.673, p = 0.003] and basal area [F (3, 5) = 19.092, p = 0.004] for the dominant tree species, jarrah, among the treatment groups (Table 3). Mean density for marri species was decreased by thinning [F (3, 5) = 6.864, p = 0.032), but the mean basal area did not differ among the thinned treatments. In all the treatment plots, *E. marginata* and *E. calophylla* were the most abundant tree species followed by *B. grandis*. However, after thinning, notch-only had higher abundance of *B. grandis* than *E. calophylla* (Figure 8), with the exception that *B. grandis* had lower basal area than *E. marginata* or *E. calophylla* in all the treatments both before and after thinning. Similar results were observed for basal area distribution among species (Figure 9).

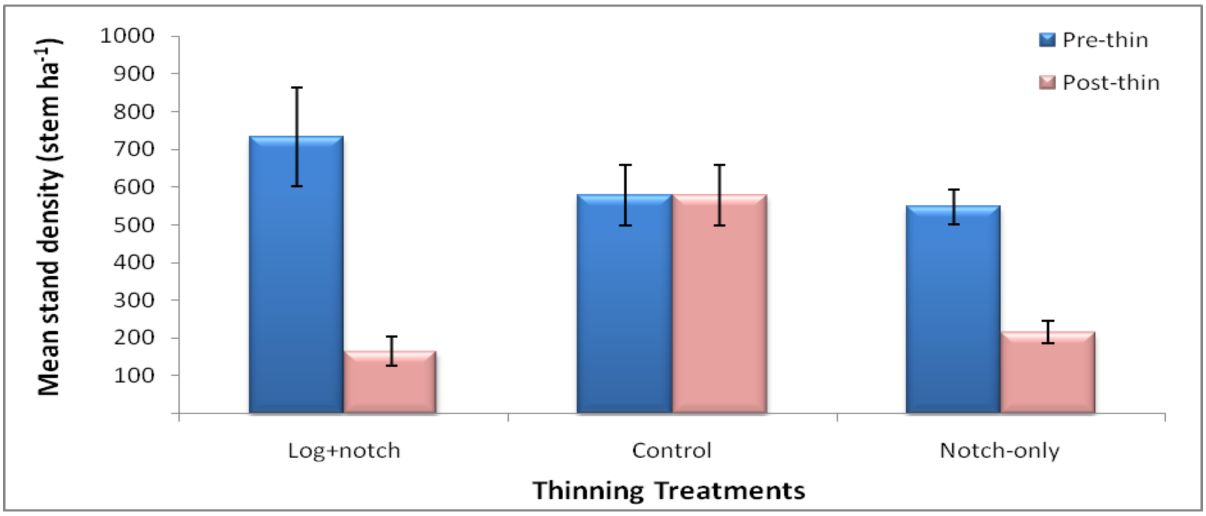


Figure 6 Pre-and post-thinning mean stand density (stem ha^{-1}) for trees with DBH ≥ 10 cm for three treatments. Values are means of three replicates. Vertical bars indicate standard errors. See Table 3 for significance of pre-thinning differences and treatment effects.

Table 3 Significance levels for analysis of variance with various attributes as the dependent variable at treatment level for stand structural parameters from the dry eucalypt forest of south-western Australia. Pre-treatment values were tested in the model as a co-variate. Where the treatment effect was significant ($p < 0.05$) the Bonferroni test was used for mean separation.

Test variables	F value	Model probability value	Probability value for covariate	Bonferroni test
Banksia density	-	ns	ns	-
Jarrah density	21.673	0.003	0.320	C>L, C>N
Marri density	6.864	0.032	0.333	C>L
Stand density total	12.428	0.009	0.288	C>L, C>N
Banksia basal area	-	ns	ns	-
Jarrah basal area	19.092	0.004	0.068	C>L, C>N
Marri basal area	2.902	0.141	0.057	
Basal area total	19.574	0.003	0.120	C>L, C>N
Stand density at 10-20 cm	8.359	0.022	0.660	C>L, C>N
Stand density at 20-30 cm	24.569	0.002	0.004	C>L, C>N
Stand density at 30-40 cm	12.711	0.009	0.016	C>L,
Stand density at 40-50 cm	7.581	0.026	0.034	C>L
Stand density at 50-60 cm	5.022	0.077	0.052	-
Stand density at 60-70 cm	10.263	0.089	0.048	-
Stand density at ≥ 70 cm	1.465	0.33	0.228	-
Basal area at 10-20 cm	9.377	0.017	0.659	C>L, C>N
Basal area at 20-30 cm	27.381	0.002	0.004	C>L, C>N
Basal area at 30-40 cm	10.120	0.015	0.028	C>L
Basal area at 40-50 cm	9.954	0.015	0.017	C>L
Basal area at 50-60 cm	2.950	0.162	0.121	-
Basal area at 60-70 cm	11.959	0.077	0.042	-
Basal area at ≥ 70 cm	0.771	0.558	0.374	-
Quadratic Mean Diameter	1.254	ns	0.779	-
Understorey tree density	0.534	0.679	0.782	-
Mid-storey tree density	33.129	0.001	0.067	C>L, C>N
Up-storey tree density	35.699	0.001	0.077	C>L, C>N

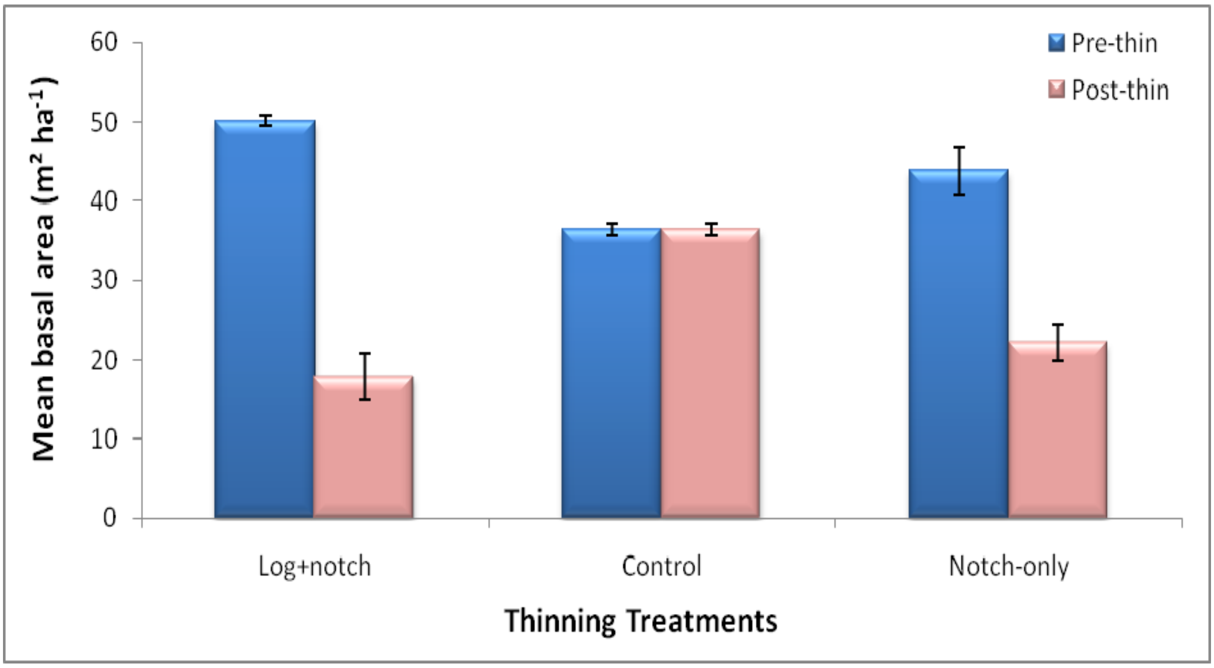


Figure 7 Pre-and post-thinning mean basal area (m² ha^{-1}) for trees with DBH ≥ 10 cm for three treatments. Values are means of three replicates. Vertical bars indicate standard errors. See Table 3 for significance of pre-thinning differences and treatment effects.

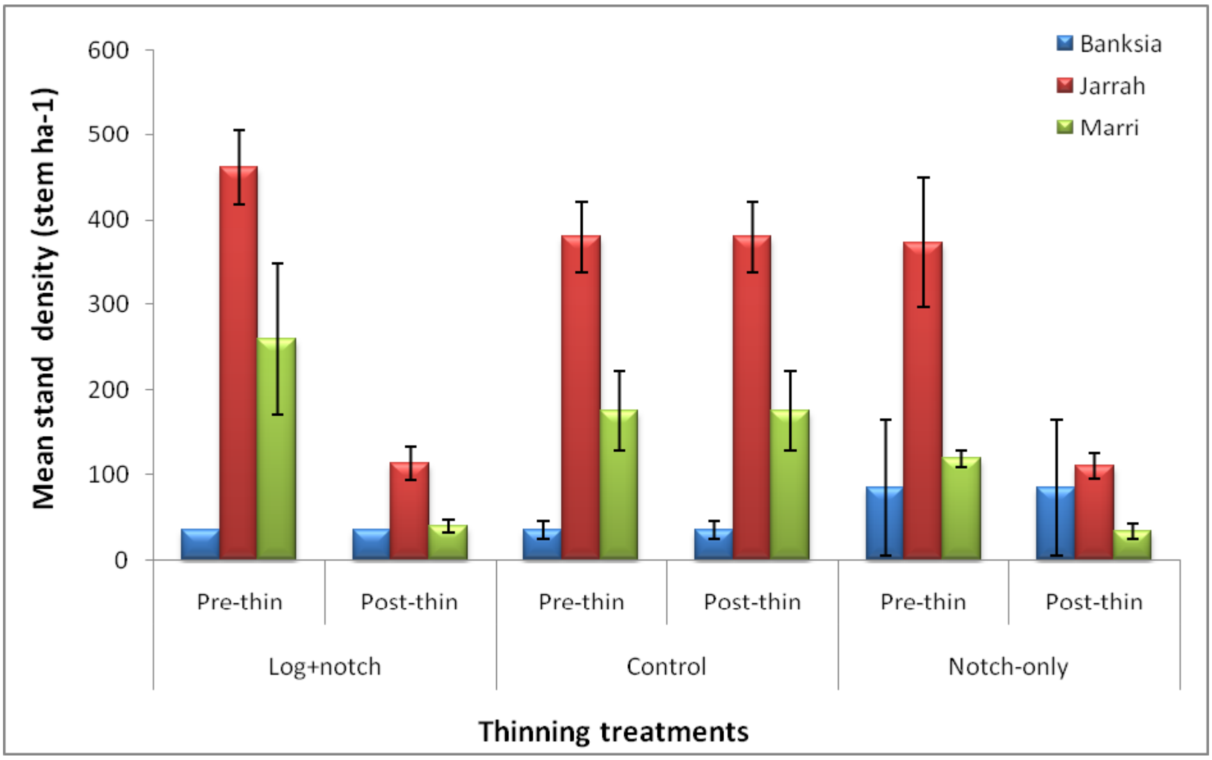


Figure **8** Pre-and post-thinning mean stand density by tree species (stem ha^{-1}) for trees with DBH ≥ 10 cm for three treatments. Values are means of three replicates. Vertical bars indicate standard errors. See Table 3 for significance of pre-thinning differences and treatment effects.

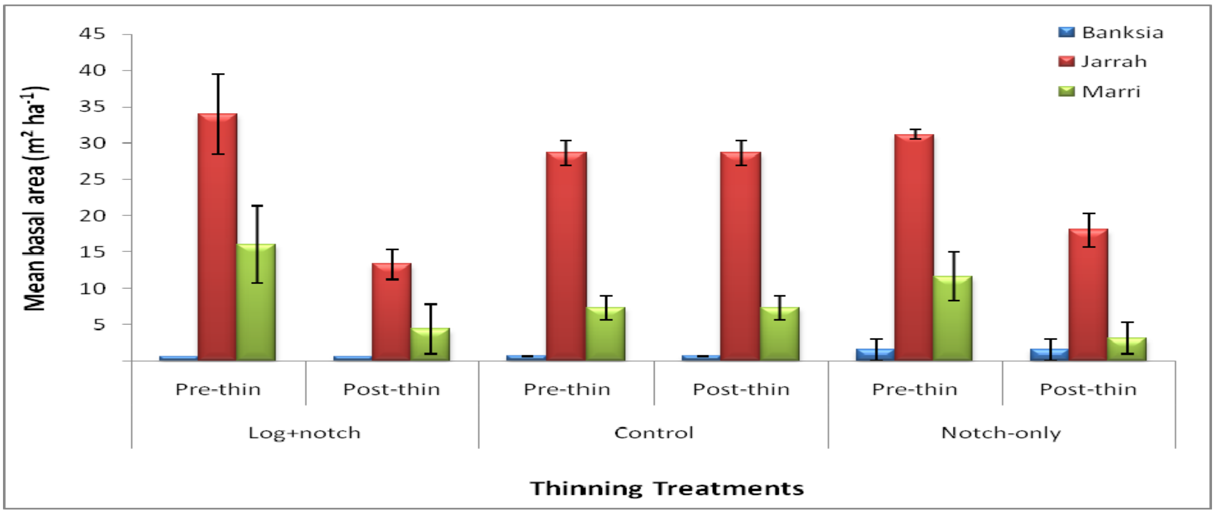


Figure **9** Pre-and post-thinning mean basal area by tree species (m^2 ha^{-1}) for trees with DBH ≥ 10 cm for three treatments. Values are means of three replicates. Vertical bars indicate standard errors. See Table 3 for significance of pre-thinning differences and treatment effects.

Pre-thinning percentage contribution of banksia, jarrah and marri trees in the total population were 5, 61 and 34 % for log+notch treatment, 6, 64 and 30 % for control treatment and 10, 68 and 22 % for notch-only treatment, respectively (Figure 10). Thinning greatly increased the percentage contribution of banksia to the post-thinning composition of the tree species. Post-thinning relative contribution of banksia, jarrah and marri were 19, 60 and 21 % for log+notch treatment and 37, 48 and 15 % for notch-only treatment (Figure 10).

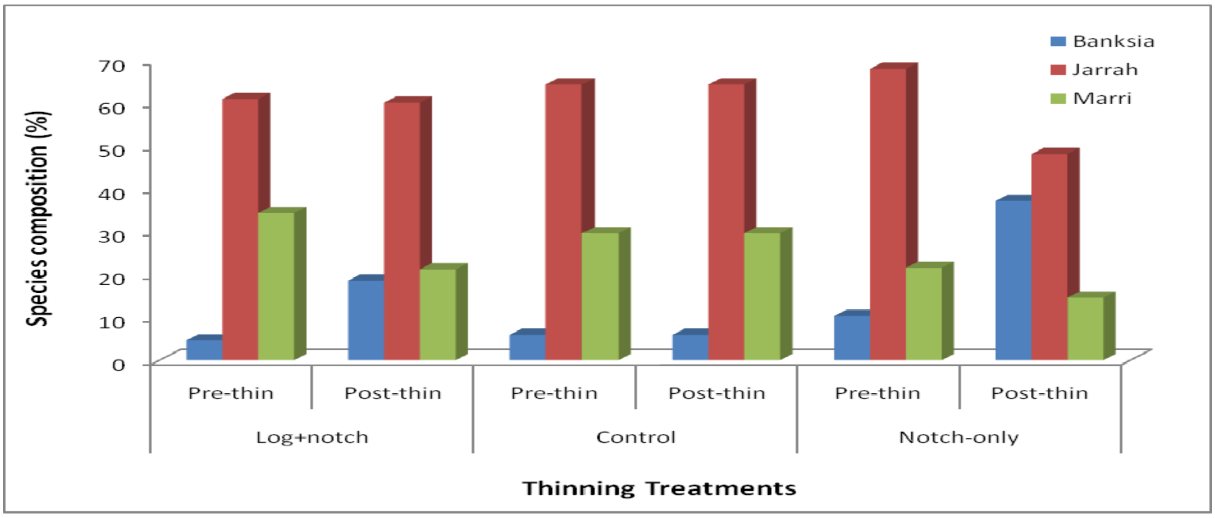


Figure 10 Pre-and post-thinning stand composition (as a % of all trees) for trees with DBH ≥ 10 cm for three treatments. Values are means of three replicates.

3.3.1.2 Stand density and basal area composition by diameter size class

Significant differences were observed among the treatments in the mean stand densities of trees in 10-20 cm DBH [$F (3, 5) = 8.359$, $p = 0.022$], 20-30 cm DBH [$F (3, 5) = 24.569$, $p = 0.002$], 30-40 cm DBH [$F (3, 5) = 12.711$, $p = 0.009$] and 40-50 cm DBH ranges [$F (3, 5) = 7.581$, $p = 0.026$] (Table 3). By contrast there was no difference among treatments in 50-60, 60-70 and ≥ 70 cm DBH classes. Pairwise comparison between the treatments with the Bonferroni test revealed that log+notch and notch-only treatments were significantly different in the stand density to control treatments for the DBH size classes 10-20 and 20-30 cm. However, there was no statistical difference between the two thinning treatments (Table 3). In addition, statistical difference was observed in the mean stand density in the 30-40 and 40-50 cm DBH classes between control and log+notch treatments, but there was no difference between the other pairs of treatments (Table 3).

Results of the ANOVA for the basal area in different diameter size classes showed variable patterns. Treatments were significantly different in the mean basal area for the diameter size classes 10-20, 20-30, 30-40 and 40-50 cm at $p < 0.05$ ([$F (3,5) = 9.377$, $p = 0.017$], [$F (3,5) = 27.381$, $p = 0.002$], [$F (3,5) = 10.120$, $p = 0.015$], and [$F (3,5) = 9.954$, $p = 0.015$], respectively) (Table 3). No statistical differences were observed among the treatments for the DBH size classes larger than 50 cm. Pairwise comparison using Bonferroni tests indicated that mean basal area for log+notch and notch-only treatment differed significantly from control treatment for 10-20 and 20-30 cm DBH size classes, but no differences were observed between them. In the diameter size classes 30-40 and 40-50 cm, mean basal area was reduced only in log+notch compared to control treatment (Figure 12).

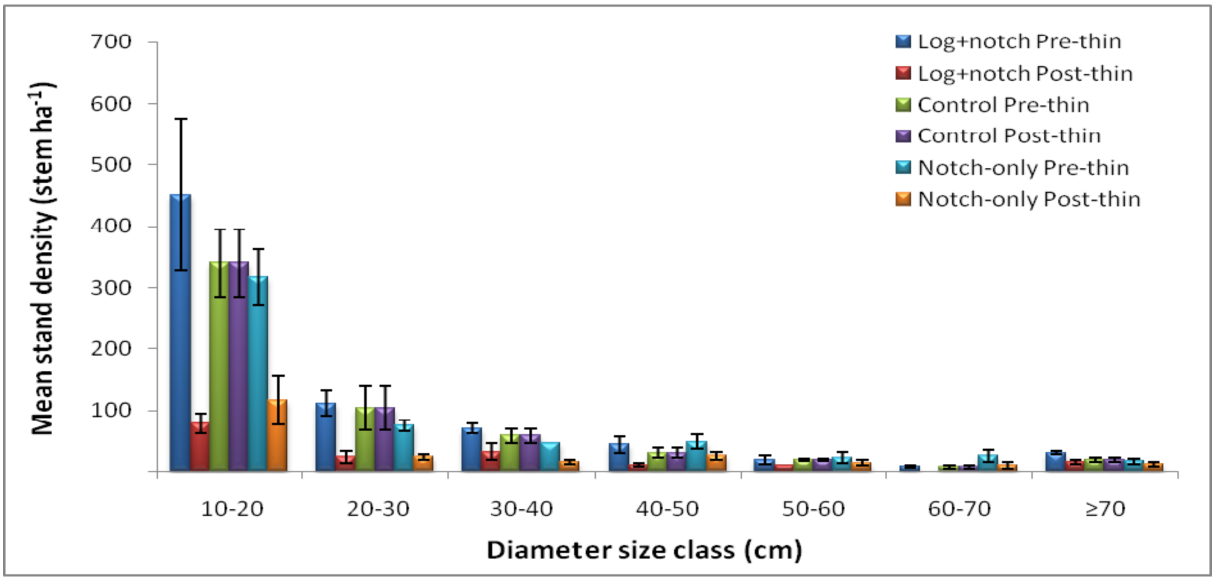


Figure 11 Pre-and post-thinning mean stand density by diameter size class for trees with DBH ≥ 10 cm for three treatments. Values are means of three replicates. Vertical bars indicate standard errors. See Table 3 for significance of pre-thinning differences and treatment effects.

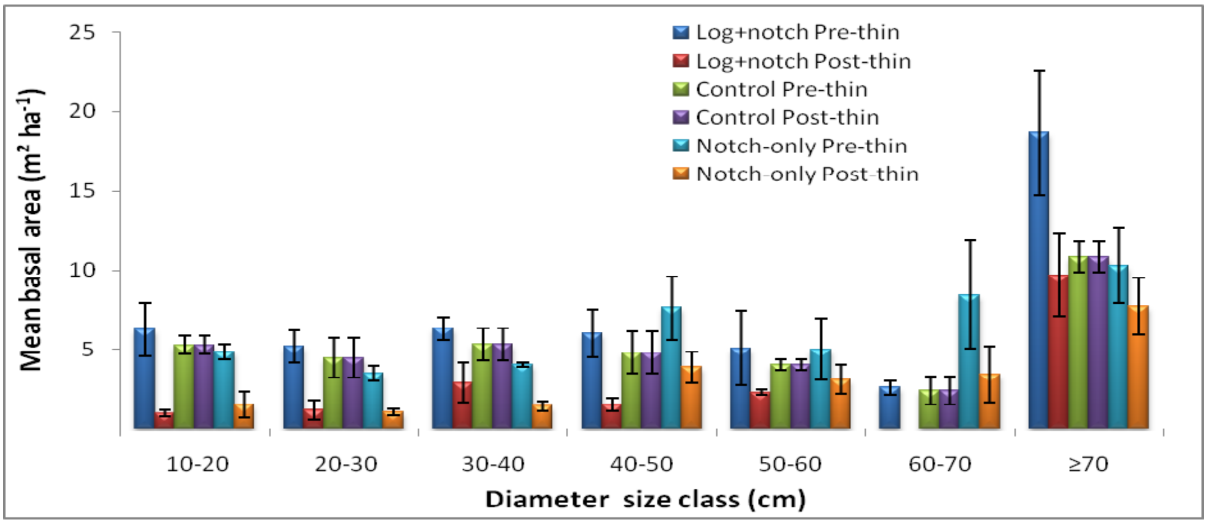


Figure 12 Pre- and post-thinning mean basal area for trees with DBH ≥ 10 cm for three treatments. Values are means of three replicates. Vertical bars indicate standard errors. See Table 3 for significance of pre-thinning differences and treatment effects.

3.3.1.3 Quadratic mean diameter (QMD)

No significant differences were observed among the treatments in the QMD after treatments [F (3, 5) = 1.254, p = 0.05] (Table 3). Within treatment comparison between pre- and post-thin QMD for log+notch treatment showed substantial difference, however, there were no difference for control and notch-only treatments (Figure 13).

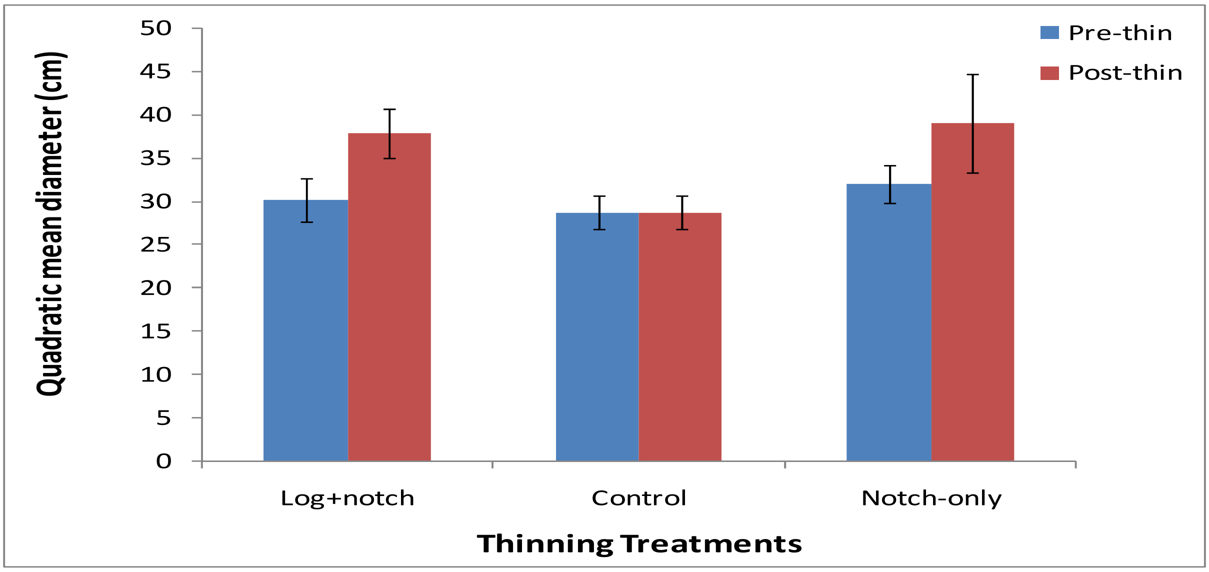


Figure 13 Pre- and post-thinning mean quadratic mean diameter for trees with DBH ≥ 10 cm for three treatments. Values are means of three replicates. Vertical bars indicate standard errors. See Table 3 for significance of pre-thinning differences and treatment effects.

3.3.1.4 Stand structure by height strata

Univariate analysis of variance for the stand density by tree height category showed significant difference among the treatment groups ($p < 0.05$) for upper storey [$F (3, 5) = 35.699$, $p = 0.001$] and mid-storey [$F (3, 5) = 33.129$, $p = 0.001$] but no difference was observed for understorey (Table 3). Pairwise comparison in the mean tree density between the treatments using Bonferroni tests demonstrated that log+notch and notch-only treatments differed significantly from the control treatment, but there was no statistical difference between them (Table 3). Before and after-thinning comparison in the mean tree density within the treatments indicated that there was significant difference in all the height strata for log+notch treatments, for the upper storey and mid-storey for notch-only treatment but there was no difference in the mean tree density in any stratum for control treatments and in understorey for notch-only treatments (Figure 14). Before and after comparison in the mean height between pre- and post-thinning plots combined together were not significantly different for all height strata (Figure 15).

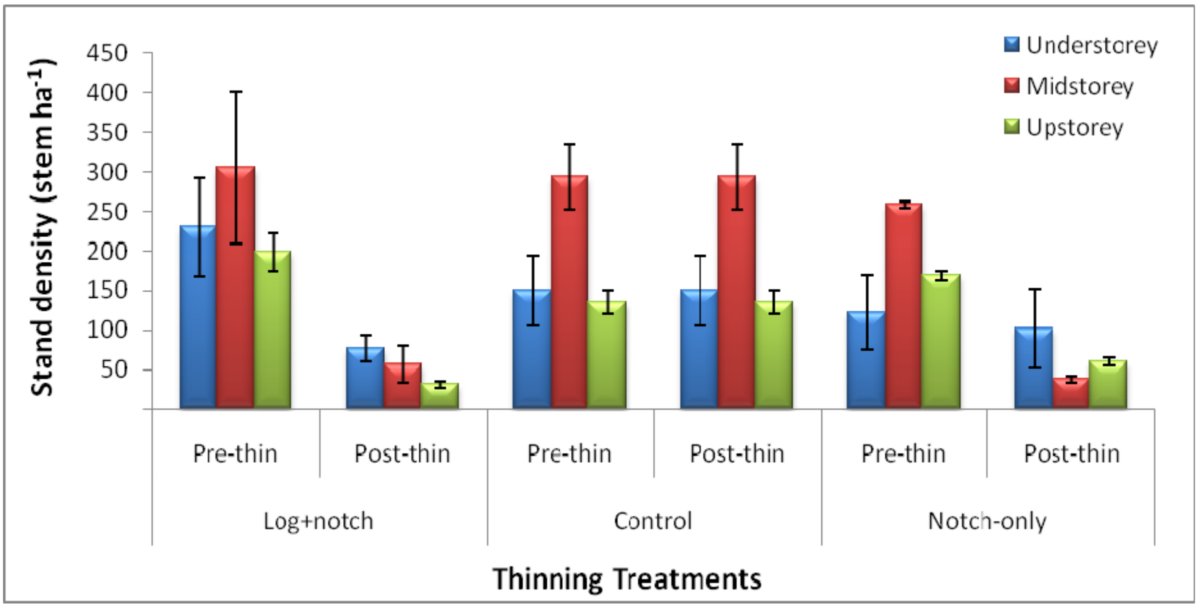


Figure 14 Pre-and post-thinning mean stand density by height strata for trees with DBH ≥ 10 cm for three treatments. Values are means of three replicates. Vertical bars indicate standard errors. See Table 3 for significance of pre-thinning differences and treatment effects. Understorey = < 10 m, mid-storey = 10-20 m and upperstorey = ≥ 20 m in height.

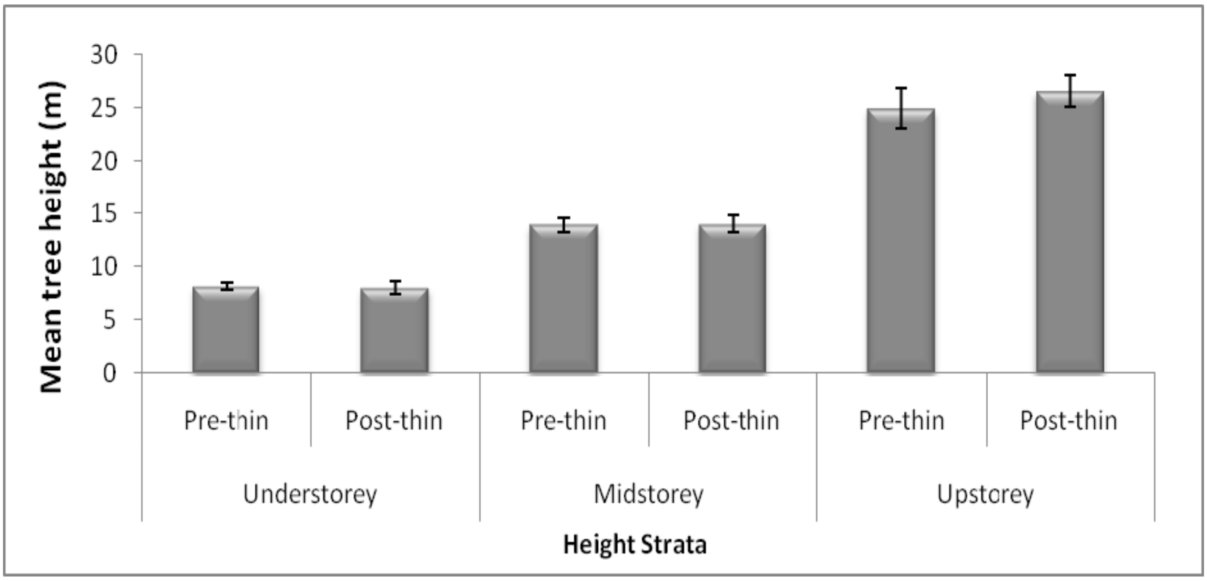


Figure 15 Mean tree height for pre- and post-thin tree stands by height strata. Values are the means of nine plots. Vertical bars indicate standard errors. See Table 3 for significance of pre-thinning differences and treatment effects. Understorey = < 10 m, mid-storey = 10-20 m and upperstorey = ≥ 20 m in height.

A regression equation developed from the measured sample data for tree diameter and the respective height of each tree revealed a strong log-linear relationship between DBH and heights for the tree stands with DBH ≥ 10 cm for jarrah and marri species (Figure 16). The regression equation generated from the measured DBH and tree height data was then applied to estimate heights of all trees in the plots. No substantial difference was observed between pre- and post-thinning mean height of each strata (Figure 15).

3.3.2 Understory species richness and ground cover

3.3.2.1 Understorey species richness

A total of 18 quadrats, 6 in each treatment, were surveyed and monitored to assess the effects of overstorey thinning on understorey species richness. A total of 40 species, 36 genera and 28 families were recorded (Table 4). Eleven plant species were recorded in the majority of the quadrats. These species are identified by asterisk in Table 4. Jarrah seedling was the most common plant species recorded during the study period followed by *Hibbertia amplexicaulis,*

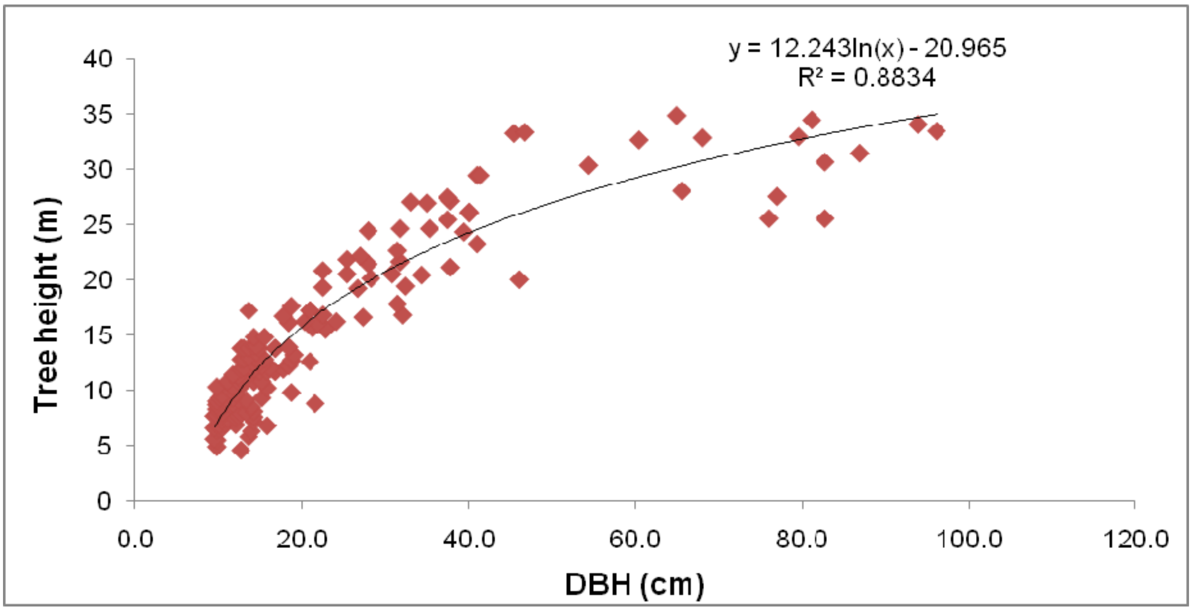


Figure 16 Relationship between DBH (cm) and tree height (m) for the whole study block at Vardi Road sub-catchment in the jarrah forest, Western Australia. Habitat trees were excluded in the sampling.

Table 4 Understorey plant species recorded during field sampling in the study quadrats in the jarrah forest, Western Australia. Plant species name followed by asterisk (*) are the most common plant species recorded in more than 61 % of the quadrats sampled.

Plant Name	Family	Type of plant	% quadrats present
Acacia urophylla	Fabaceae	Shrub	6
Adenanthos barbiger	Proteaceae	Shrub	6
Banksia grandis	Proteaceae	Tree	33
Boronia heterophylla	Rutaceae	Shrub	22
Bossiaea ornate	Papilionaceae	Shrub	22
Clematis microphylla	Ranunculaceae	Herb	50
*Clematis pubescens**	Ranunculaceae	Herb	78
Conostylis setose	Haemodoraceae	Herb	6
*Corymbia calophylla**	Myrtaceae	Tree	61
*Corynotheca micrantha**	Anthericaceae	Herb	78
Desmocladous fasciculatus	Restionaceae	Herb	44
Drosera bulbosa	Droceraceae	Herb	11
*Eucalyptus marginata**	Myrtaceae	Tree	100
Gompholobium marginatum	Papilionaceae	Shrub	17
*Hibbertia amplexicaulis**	Dilleniaceae	Shrub	89
*Hibbertia commutata**	Dilleniaceae	Shrub	83
Hovea chorizemifolia	Papilionaceae	Shrub	17
Laginifera huegelii	Asteraceae	Herb	28
Leucopogon verticillatus	Lamiaceae	Shrub	6
*Lomandra preissi**	Dasypogonaceae	Herb	67
Macrozamia riedlei	Zamiaceae	Tree	22
Mirbelia dialatata	Fabaceae	Shrub	6
*Neurachne alopecuroidea**	Poaceae	Herb	78
Opercularia hispidula	Rubiaceae	Herb	17
Oxalis corniculata	Oxalidaceae	Herb	11

Patersonia babianoides	Iridaceae	Herb	11
Pentapeltis peltigera	Apiaceae	Herb	33
Persoonia longifolia	Proteaceae	Tree	22
Pimelia species	Thymelaeaceae	Shrub	6
*Pteridium esculentum**	Dennstaedtiaceae	Herb	78
Scaevola pilosa	Goodeniaceae	Herb	50
Scaevola species	Goodeniaceae	Herb	6
Styphelia tenuiflora	Epacridaceae	Shrub	17
*Thomasia foliosa**	Sterculiaceae	Shrub	100
Thysanotus multiflorus	Anthericaceae	Herb	11
*Trichocline spathulata**	Asteraceae	Herb	67
Unidentified	-	Herb	11
Xanthorrhoea gracilis	Xanthorrohoeaceae	Tree	11
Xanthorrhoea preissii	Xanthorrohoeaceae	Tree	22
Xanthosia species	Apiaceae	Herb	11

Hibbertia commutata, Corynotheca micrantha, *Clematis pubescen, Neurachne alopecuroidea* and *Pteridium esculentum* whilst *Acacia urophylla, Adenanthos barbiger, Conostylis setosa Pimelea species, Leucopogon verticillatus, Mirbelia dialatata* and *Scaevola species* were the least common species recorded during the study period (Table 4). A summary of the average species richness for each treatment for the 5 consecutive survey periods starting before thinning on August 2009 and continuing through post-thinning until September 2010 are presented below in Table 5. Seasonal changes in species richness and ground cover are also presented in Figure 17 and Figure 19, respectively.

There was no statistically significant difference in mean understorey species richness for the period monitored. Pre- and post-thinning comparison in species richness did not show significant differences either. However, there were seasonal changes in the numbers of species recorded (Figure 17). The maximum species richness was observed in the September 2010 survey *viz*: 18 species in quadrat 1 of log+notch treatment (Figure 17 (a)), 16 in quadrat 3 of control treatment (Figure 17 (b)) and 16 in quadrats 1 and 3 of notch-only treatment (Figure 17 (c)). The lowest species richness recorded was 12 at quadrats 2, 5 and 6 in the log+notch treatment (Figure 17 (a)), 10 at quadrats 4 and 5 in control treatment (Figure 17 (b)) and 9 at quadrat 6 in notch-only treatment (Figure 17 (c)). Results of the species-area plot for the understorey sampling revealed that the number of sampling quadrats (2×2 m) may need to be sixteen or more (64 m^2) to pick up near highest number of understorey species present in the study site (Figure 18). However, 6 quadrats (24 m^2) in each treatment obtained approximately 80 % of the species recorded in this study (Figure 18).

Table 5 Summary of average species richness for three thinning treatment groups between August 2009 (pre-thinning) and September 2010. Values are means of six quadrats in each of three replicate plots. Note: August 2009 was pre-thinning, while the December 2009 to September 2010 measurements represent post-thinning data. Differences in average species richness were not significant among measurement periods.

	Average species richness± S.E.				
Treatment	Aug.2009	Dec.2009	Mar.2010	Jun.2010	Sept.2010
Log+notch	13.2±0.98	11.8±1.01	11.7±0.62	13.3±0.72	14.2±1.05
Control	11.3±0.76	10.8±0.95	11.3±0.88	12.8±0.83	12.3±1.05
Notch-only	12.8±1.22	11.3±1.23	12.5±1.23	13.8±0.48	13.3±1.09

3.3.2.2 Understorey ground cover

There was no significant difference in the mean understorey ground cover among treatments; however there was a significant difference in the mean understorey ground cover at site level at $p < 0.05$ [$F(3,14) = 47.781$, $p = 0.000$]. There was a slight decline in the percentage ground cover in all the treatment groups during December and March sampling (6.3 % for log+notch treatment, 6.2 % for control and 0.2 % for notch-only treatment) (Table 6).

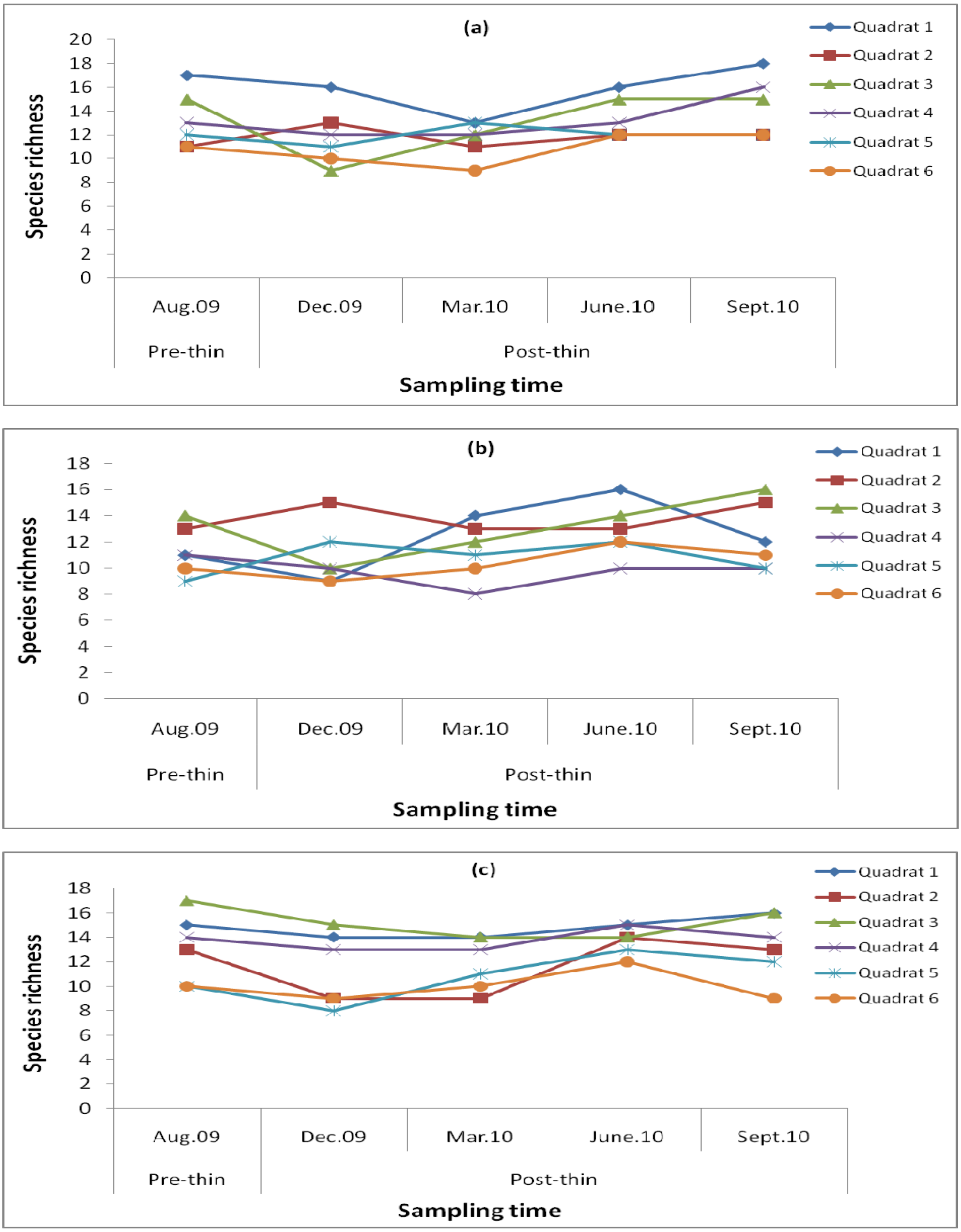


Figure 17 Species richness recorded at individual quadrats in log+notch (a), control (b) and notch-only treatment (c) from Aug. 2009 to Sept. 2010. Vertical axis indicates the number of species recorded.

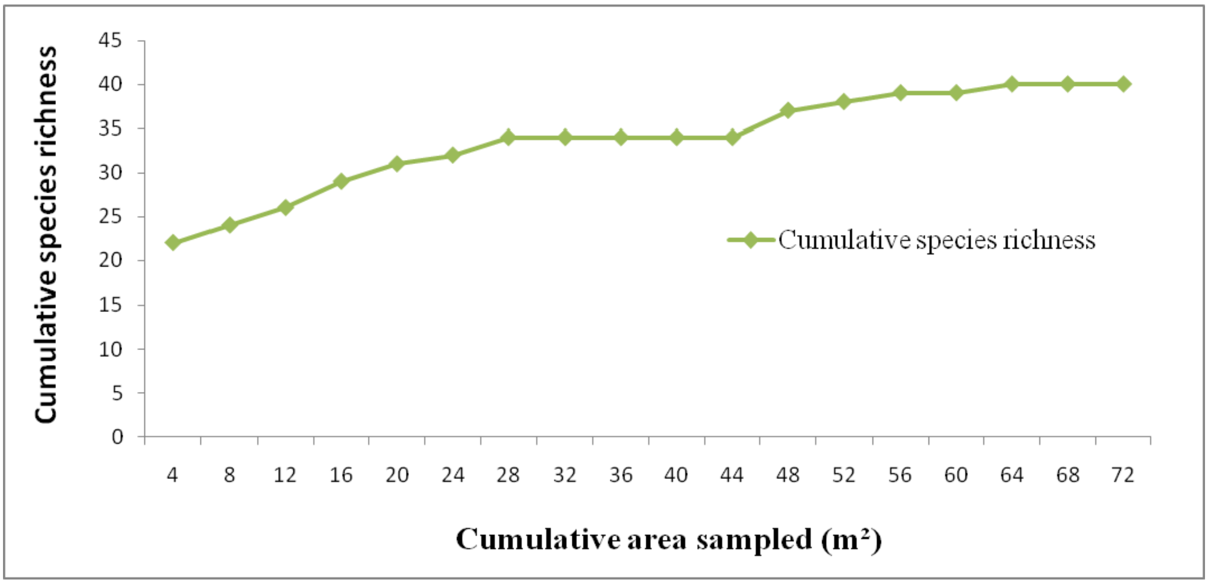
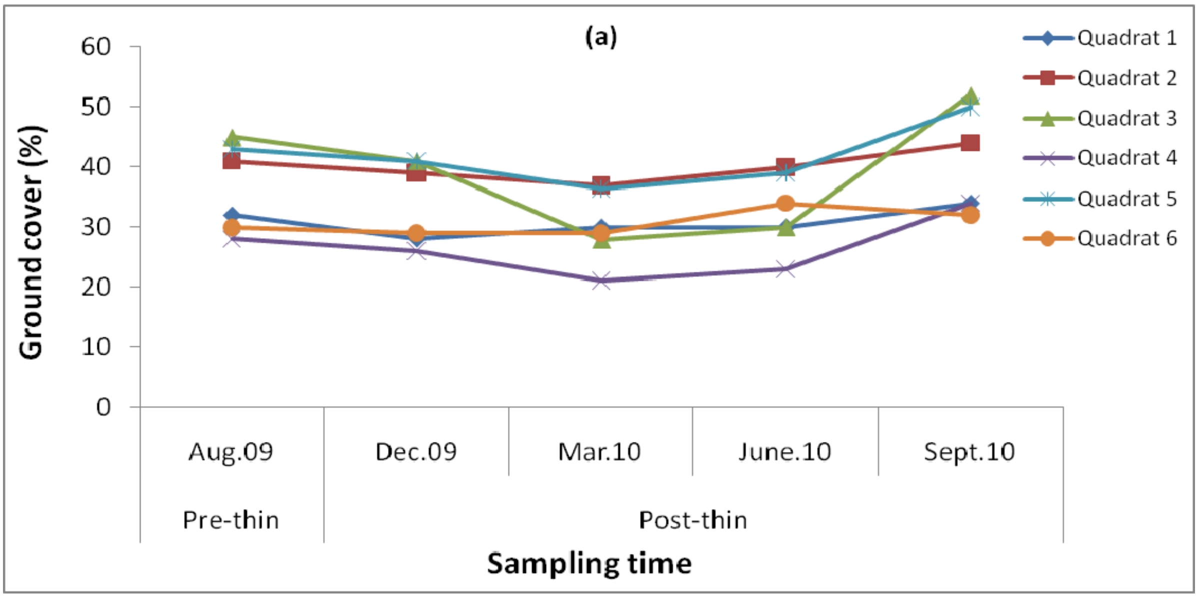


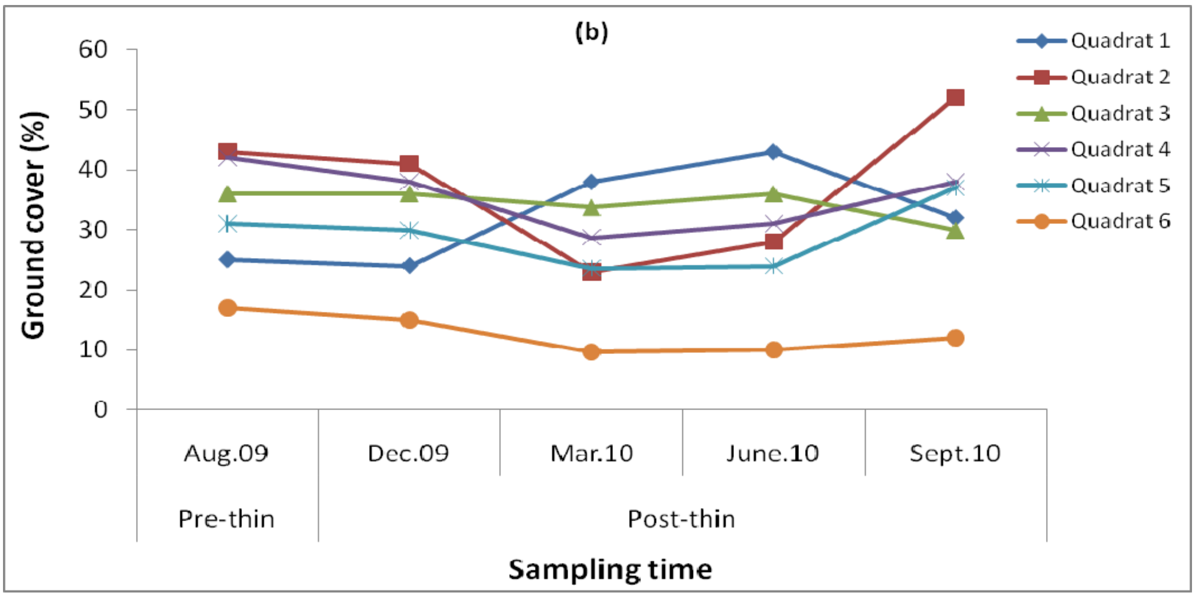
Figure 18 Species-area plot showing the relationship between cumulative area of the sampling quadrats 18 quadrates, each 4 m^2 were sampled in total and the cumulative understorey species richness. Vertical axis indicates the number of species recorded.

However, comparison between pre- and post thinning ground cover value within the treatment showed a slight increase (4.5 % in log+notch, > 1 % in control and 3.2 % in notch-only treatments) (Table 6). Maximum understorey ground cover assessed on September 2010 was 52 % in quadrat 3 in the log+notch treatment (Figure 19 (a)), 52 % in quadrat 2 in the control treatment (Figure 19 (b)) and 40 % in quadrat 1 in the notch-only treatment (Figure 19 (c)). The lowest understorey ground cover recorded was 32 % at quadrat 6 in the log+notch treatment (Figure 19 (a)) and 12 % at quadrat 6 and 3 in control and notch-only treatments, respectively (Figure 19 (b) and 19 (c)). Before and after comparison in the ground cover at quadrat level did not show a significant change ($p > 0.05$). However, in some quadrats there was an increase in cover value- for example, at quadrat 3 and 5 in log+notch treatment (Figure 19 (a)) and quadrat 2 and 5 in control treatment (Figure 19 (b)).

Table 6 Summary of average ground cover for three thinning treatment groups. Values are means of six quadrats in each of three replicate plots. Note: August 2009 was pre-thinning and rest of others is post-thinning data.

	Average ground cover± S.E.				
Treatment	Aug.2009	Dec.2009	Mar.2010	Jun.2010	Sept.2010
Log+notch	36.5±3.00	34.0±2.88	30.2±2.39	32.7±2.60	41.0±3.61
Control	32.3±4.13	30.7±4.00	26.1±4.02	28.7±4.60	33.5±5.33
Notch Only	17.3±3.67	16.2±3.20	17.5±3.68	19.3±5.18	20.5±4.03





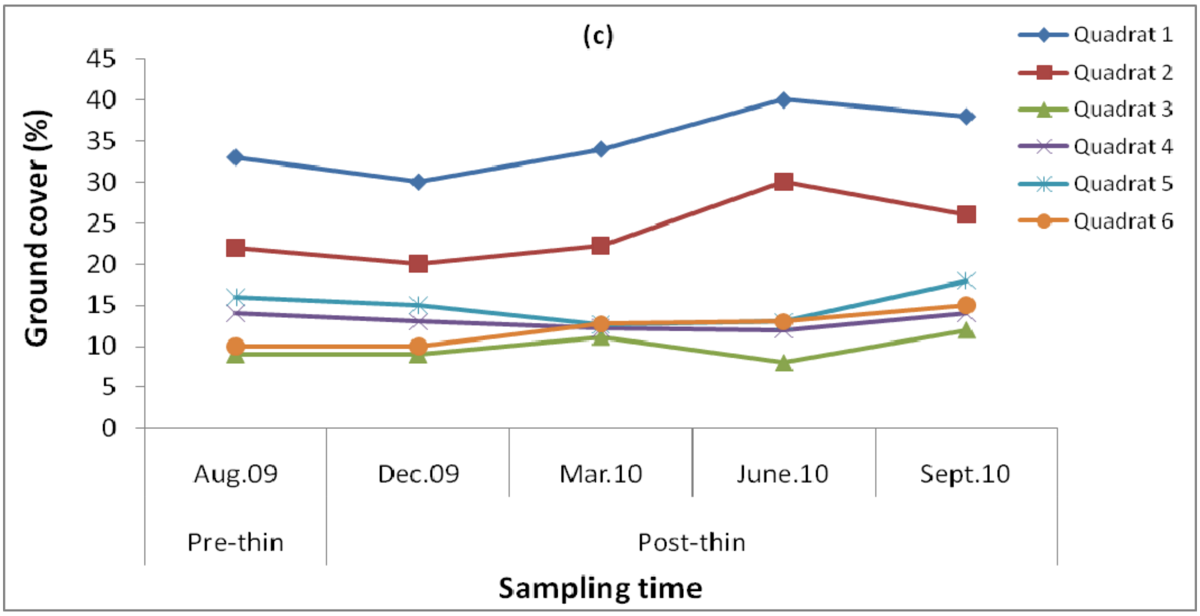


Figure 19 Understorey ground cover (%) assessed at individual quadrats in log+notch (a), control (b) and notch-only treatment (c) from Aug. 2009 to Sept. 2010.

3.4 Discussion

3.4.1 Forest structure and composition

Methods and intensity of tree thinning had a significant impact on post-thinning structure and composition of the jarrah forest stand in this study. Both stand density and basal area were significantly decreased by thinning relative to the pre-thinning forest stands (Table 3). Post-thinning densities of the jarrah and marri species were significantly lower than the pre-thinning condition as thinning was directed towards reducing density of these two species (Department of Environment and Conservation 2007). By contrast, no difference was observed in the density of banksia because all the banksia trees were intended in the prescription to be untreated (Department of Environment and Conservation 2007). Basal area for jarrah was significantly reduced. However, there was no significant reduction in the basal area for marri as they were less abundant prior to thinning and therefore comparatively less affected by thinning treatment (Table 3). This result appeared to be consistent with the

Silvicultural guidelines developed for thinning in Wungong catchments (Department of Environment and Conservation 2007). While the density of banksia did not change, it increased in prevalence as a % of the retained tree stand.

3.4.1.1 Stand density and basal area composition by tree species

Pre-thinning site differences were significant for stand density and tree species composition among the plots (Table 3). Without including the pre-thinning data as a co-variate, the level of heterogeneity may have produced misleading interpretation of the thinning effects. Such variability in the forest may be attributed to topography, availability of soil moisture, degree of soil leaching and resulting acidity; and soil fertility (Bell and Heddle 1989). Designing smaller plots may ensure greater uniformity of the forest stands, but may compromise the reliability of other data such as species richness and ground cover of the understory component. Before thinning, tree species distribution in the treatment plots were non-random at small scale (plot scale) for marri and banksia although jarrah was distributed at random similar to the results observed for the jarrah forest (Abbott 1984). Higher densities of banksia were found in plot 6 and 9 while for marri they were higher in plots 7 and 4. A comparative study of spatial pattern, structure and tree composition between virgin and cut-over jarrah forest of Western Australia demonstrated that most populations of the species were aggregated at small scale comparable to the area of plots in the present study (ranging from 0.8 ha to 1.28 ha) but random at large scale (Abbott 1984). In this study, *E. marginata* was the most widely distributed species of the jarrah forest- occupying approximately 65 % of the total population- followed by *C. calophylla* (30 %) and *B. grandis* (5 %).

The upper slopes and ridges in the northern jarrah forest where there was a the dominance of jarrah and marri in the overstorey and banksia in the understorey were classified as Site-vegetation type P (Havel 1975a, b). Other indicator understorey species such as *Hovea* *chorizemifolia, Persoonia longifolia, Leucopogon verticillatus* and *Styphelia tenuiflora* were also present. In this study, pre-thinning percentage contribution of banksia, jarrah and marri trees in the total population for all treatment groups were approximately similar i.e. 5, 61 and 34 % for log+notch treatment, 6, 64 and 30 % for control treatment and 10, 68 and 22 % for notch-only treatment, respectively (see Figure 10). Thinning greatly increased the relative abundance of banksia to the post-thinning composition of the tree species. Post-thinning relative contribution of banksia, jarrah and marri were 19, 60 and 21 % for log+notch treatment and 37, 48 and 15 % for notch-only treatment (see Figure 10). This change in the species composition due to thinning may be expected to have significant effects on the species composition of the forests in the future. For example, the greater proportions of banksia in the thinned forest may have longer term consequences for forest health given its high susceptibility to *Phytophthora cinnomomi*.

3.4.1.2 Stand density and basal area composition by diameter size class

Highly significant differences were observed in post-thinning mean stand densities for the 20-30 and 30-40 cm DBH size classes (Table 3) indicating that thinning was targeted at reducing medium-sized trees. The rationale for removing these trees is consistent with decreased vegetation water use in the thinned forest because this class of trees have a higher proportion of sapwood to non-sapwood area as compared to larger trees (Macfarlane and Silberstein 2009). This means that for each unit of basal area removed by tree thinning, the decrease in water use will be greatest if targeting the trees with high sapwood to non-sapwood area. Large numbers of trees in the 10-20 cm diameter size class and very few in the larger diameter size classes in the pre-thinning inventory suggest a regrowth stand that has developed from progressive reduction in tree-sizes suitable for timber. Even though the site has not been logged for about 60 years, large trees were still relatively uncommon.

Furthermore, a large number of coppice stems developed from cut-stumps and evidence of decomposed logs piled on the site (see Chapter 4) support the claims that the forest has developed as a regrowth stand after a series of timber harvests.

Comparison of spatial pattern, structure and tree composition between virgin and cut-over jarrah forest near Jarrahdale and Dwellingup also demonstrated that most populations of the species are aggregated at small scale but random at large scales (Abbott 1984). This aggregation is probably due to non-random seed fall. The species included in Abbott (1984) study were *E. marginata, E. calophylla* in the overstorey and *B. grandis, A. fraseriana, P. longifolia* and *P. elliptica* in the understorey. In this study, field observation suggests that distribution of jarrah trees was random, however less random for marri but strongly localised for banksia. This aggregation may be related to the heterogeneity in physical and chemical properties of the soil or the composition and structure of the litter layer, competitive interaction between species, the pattern of reproduction, recruitment of seedlings, or the development of advanced growth. Localised distribution of banksia species may be attributed to the inefficient dispersal of seeds by wind (Abbott, unpublished). Alternatively, localised distribution of *Banksia grandis* in the understorey was observed indicating low fertility status of the soil (Havel 1975a, b).

3.4.1.3 Quadratic mean diameter (QMD)

There was no significant difference in the quadratic mean diameter among the treatments. This may be due to the fact that the trees were marked for thinning with a view to achieve natural composition of the forest stock both in terms of stand density and the distribution of stems at different diameter size-classes. QMD is commonly used in silviculture research data summaries and reports. Virtually all normal yield tables prepared in the United States in the period from around 1920 through the mid- 1960s use quadratic mean diameter (Schnur 1937, McArdle et al. 1961, Barnes 1962), however in Australian forest studies stand density and basal area are commonly used (Stoneman and Whitford 1995, Stoneman et al. 1996). The greater sensitivity of BA and stand density measures to thinning treatments in the present study suggest that these parameters rather than QMD are more valuable for assessing change in stand structure.

3.4.1.4 Stand structure by height strata

Significant decrease in the mean stand density in the mid-storey and upper storey was observed for log+notch and notch-only treatments, but the two thinning treatments had similar effects even though the retained BA of log+notch treatment was greater and the intensity of thinning was also greater (Table 3). There was no significant difference in the pre-thinning mean stand density in each height strata among the treatment groups. Mean height of trees in each height strata were not affected by thinning suggesting that the trees selected for removal were representative of the overall stand in terms of their height. The largest percentage of trees was in the mid-storey, which might be due to the fact that trees with <10 cm DBH were excluded in this study. In the post-thinning treatment groups, mean density of trees in the understory layer was highest followed by mid-story and upper storey, which is consistent with the fact that understory trees were little affected by thinning treatment.

In a previous study of thinning effects on forest structure of jarrah forest of Western Australia, no attempts were made to classify individual trees by height strata (Abbott 1984). In a survey of the flora and vegetation of the Wungong catchment (Mattiske Consulting Pty Ltd. 2007), mean height for individual species for four upland transects were presented for trees (defined as 130 cm or taller) and seedlings (shorter than 130 cm), but the results were not classified into different height strata. Mean height of trees for all species combined were

549 ± 18.0, 716 ± 24.7, 692 ± 40.6; and 757± 83 cm for transects 1-1, 1-2, 2-1; and 2-2, respectively. Distribution of stems for each tree species by diameter class measured at breast height (DBH) was also presented. In all transects, trees with stems measuring between 5 and 15 cm DBH generally comprised about 50 % of the trees assessed. Stems <5 cm and between 15 and 25 cm were the next most commonly recorded. The frequency of stems between 5 and 15 cm is symptomatic of this age of stand. Very few trees with DBH >45 cm were recorded in each transect: these are supposed to be left as seed trees. In the present study, attempts were made to classify tree stands in to three different height classes as described above (Figure 14 and 15). A substantial number of trees in the midstorey height strata in the entire treatment group were observed in the pre-thin stands followed by understorey and upstorey. Thinning altered the composition of tree stands in the post-thin stands by reducing the number of stems in the understorey and midstorey. It is predicted that reduction in the numbers of trees in the understorey and midstorey, i.e. decrease in the tree stems with higher sapwood area, will increase the amount of run-off entering the catchment (Macfarlane and Silberstein 2009). Moreover these changes in the structure and composition of tree stands are expected to increase available soil moisture and nutrients to the retained trees consequently influencing the post-thinning ecological processes such as recruitment, regeneration and survival.

3.4.2 Understory species richness and ground cover

3.4.2.1 Understorey species richness

No statistically significant difference in the mean understory species richness was observed among the treatments for the periods monitored. This may be due to the fact that all the sampling quadrats were not directly disturbed by logging operation and log removal, however; the sampling quadrats were established randomly within the larger sub-plots. Direct physical disturbance to the soil and understorey vegetation within 50×40 m sampling sub-plots by the machinery during log removal for log+notch plots were also recorded to be minimal, ranging from 0-12 % of the area of logged plots (approximately 10 % for plot 1 and 12 % for plot 5 with no disturbance for plot 7). In the present experiment, log removal was done with light weight machinery; hence the extent of physical disturbance was less than would occur in the routine forest thinning operation. Estimates of areas physically disturbed in these operations vary depending up on the type of machines used, moisture condition of the soils and the types and intensities of logging.

Substantial seasonal reduction in the understorey species for the majority of the quadrats in all treatment groups were observed in December, 2009 and March 2010 relative to other times of the year. Species not observed during the dry period reappeared in July 2010 and September 2010 sampling in thinned as well as unthinned plots. This seasonal disappearance of some herbaceous species such as *Opercularia hispidula, Conostylis setosa and Desmocladous fasciculatus* might be related to the availability of soil moisture. During the dry season, lack of available moisture may have resulted in disappearance of these herbaceous species. Study conducted in jarrah forest of south-west Western Australia on the short-term impacts of logging on understorey vegetation also reported no difference in the native plant species richness between unlogged and adjacent logged patches at coupe scale; however mean number of species per 1 m² was 20-30 % higher in the unlogged buffers than the logged patches (Burrows et al. 2001). Two silvicultural treatments, i.e., gap cutting and shelterwood cutting, were evaluated 4 years after logging. The soil was mechanically disturbed by heavy machinery during felling, log extraction and stockpiling operations, which covered 60-80 % of the area of logged coupes. Abundance of native perennial herbs, sedges and woody shrubs was significantly lower in the logged coupes, but the abundance of short lived herbs and introduced weeds was higher. Burrows *et al.* (2001) outlined several possible causes of reduction in the abundance of these taxa following logging such as soil disturbance and damage, physical damage to the vegetation, silvicultural burning, post-logging herbivory; and combinations of these factors. While it was possible that a combination of factors contributed to the reduction, they believed that soil disturbance and physical damage to the vegetation during and after logging operations were the primary causes, while localised heating of the soil during the silvicultural burn also contributed. Result of present study for understorey species richness and ground cover appeared to be consistent with the study conducted in 32-year-old, even-aged regrowth stands of silvertop ash (*Eucalyptus sieberi*) regenerated after a wildfire (Bauhus et al. 2001), although there were significant differences in age after thinning, thinning intensity and study site characteristics between studies. Six years after commercial thinning of the forest stands by a basal area reduction of 50 % with coppice control, there were no significant differences in species richness between thinned and unthinned stands but thinning promoted the abundance of herbaceous species. A similar study was conducted by Mattiske Consulting Pty Ltd (2004) in the spring months of 2003 at Inglehope forest block. It was first thinned in 1964 (at age 40 years) and received a second thinning and fertilization in 1986 (stand basal area under bark: 5.5, 10.9, 16.4, 22.4, and 28.5 m^2 ha^{-1}, and two fertilizer treatments: unfertilized and fertilized with 400 kg ha^{-1} N and 229 kg ha^{-1} P, by three replicate plots). In both the 1964 and 1986 thinning, the smaller and the slower-growing trees were removed (Smith 1962). Marked differences in the floristic composition of the communities were observed between the treatment plots (Mattiske Consulting Pty Ltd. 2004). However, the pattern of occurrence of some of the species was not consistent with different thinning regimes, suggesting that the local site conditions may be more significant in determining the floristic composition than the thinning regimes. These differences might be due to local site conditions such as climate, landforms and soils that play a significant role in determining the distribution of plant communities in this area (Havel

1975a, b, Heddle et al. 1980, Mattiske and Havel 1989). Furthermore the local variation in some of the parameters measured, such as species richness, density and foliage cover indicated that the issues are more complex than initially thought and that the variation across the site may exceed the variation resulting from the thinning treatments. At this point, it is important to recognize that the trial was not established to investigate the impacts of different thinning regimes on the understorey species and therefore no initial assessments were undertaken on the understorey at the time of the trial establishment. Due to lack of pre-thinning data to establish baseline condition, Mattiske Consulting Pty Ltd (2004) reported that it was difficult to assess whether the differences in the understorey species richness, density and foliage cover were attributed to pre-thinning site variability or the method and intensity of thinning. A significant advantage of the present study for any future assessment of thinning effects on species richness and diversity is the baseline data that have been recorded.

3.4.2.2 Understorey species richness

Understorey ground cover in this study did not differ significantly between the treatments, but pre- and post-thinning comparison of understorey ground cover for August 2009 and September 2010 at quadrat level showed a slight increase in cover value for some quadrats. An increase in herb cover was observed shortly after thinning in *Eucalyptus sieberi* regrowth stands in East Gippsland (Kutt *et al.*, unpublished) suggesting that herbs, which usually occur in the lower strata of the understorey may be light-limited in these forests. However, in the present study, light may not be the primary limiting factor rather it might be either water or the nutrients or both (Stoneman et al. 1995, Stoneman and Whitford 1995). Differential treatment impacts on limiting resources, particularly underground resources, could explain the variable understorey response to thinning (Coomes and Grubb 2000). In a review of ecological literature, Coomes and Grubb (2000) found that light was the primary limiting resources where moisture and nutrients are readily available, while underground resources such as water and nutrients were more often considered to be limiting resources in dry forests. Available nitrogen and moisture are typically the limiting resources in the regionally dry and relatively infertile forest systems (Riegel et al. 1992). 2010 was an extremely dry year with an annual rainfall of 614.8 mm (average rainfall recorded for the Jarrahdale weather station), which is the lowest in the last three decades: hence the effects on understorey species richness and ground cover may not be comparable with the results obtained during an average rainfall year. Insignificant changes in the understorey species richness and ground cover may be attributed to the lack of readily available moisture in the upper layer of the soil horizon, so the expected changes in the understorey components may not be similar to the results obtained from other similar studies conducted from an average rainfall years.

Building on the present findings, there are two considerations for further generalization of the results. Firstly, the present study site does not represent the full range of jarrah forest site types, as it was only conducted in upper slopes of the jarrah forest dominated by jarrah and marri in the overstorey and banksia in the understorey (site-vegetation type S and P) (Havel 1975a). Secondly, the present study does not provide a long-term trajectory of the vegetation composition. In this study, only 6 quadrats in each treatment (24 m^2 in total) were monitored for understorey species richness and ground cover in areas with canopy removed, and no quadrats were located in the areas directly disturbed by the logging operation. Hence the effects of direct disturbance on understorey have not been assessed, although the disturbed area comprised up to 10 % of the total logged area. Total area sampled per plot was not large enough to obtain maximum species present in the study area as approximately only about 80 % of the species were recorded. Also, the duration of monitoring since thinning may not be sufficient to understand the post-thinning ecological processes in the longer term or at a larger scale.

There remains uncertainty about effects of changes in the overstorey structure and composition of the forest on understorey species richness, abundance, and ground cover in the long-term. This needs on-going investigation. Minimum physical disturbance observed in this study will not have more than localized impact on important soil-borne organisms and on processes such as nutrient cycling, hydrology and soil erosion unlike the thinning and burning studies that have reported the negative consequences due to widespread physical disturbance (Metlen and Fiedler 2006). However, significant levels of logging and amounts of thinning residues produced from logging and notching of trees are expected to affect nutrient recycling processes (Qiu et al. 2010). As in many Australian forest studies, the soil is the major nutrient store, followed by the above-ground biomass and litter (Hingston et al. 1981). As most of the jarrah forest has been selectively logged and periodically burnt, the current nutrient store in trees depends on how much timber has been removed and the rate of growth of the live biomass (Hingston et al. 1981). Stores in shrubs and litter depend largely on how long it is since the forest was burnt and the intensity of the fire (Hingston et al. 1981). Differences in the nutrient concentrations and the biomass between forest ecosystems can be attributed partly to the relative proportions of wood and leaves. For jarrah trees, the proportion of leaves is small (approximately 2% of the total tree biomass) (Hingston et al. 1981).

To explore the longer term impacts of thinning on structural components of the jarrah forest ecosystem, a series of further studies especially focussing on both overstorey and understorey components is essential. As most of the forests of Western Australia have been reported to be overstocked with unproductive tree components, thinning of the forest stands combined with periodic controlled burning can play an important role in reducing the stocking density on the one hand, and on the other hand it may help to promote forest biodiversity in long term (Burrows 2008). Thinning combined with controlled burning of the forest stand not only reduces the risks of wild fire, but also helps to promote growth and development of the retained crop trees (Burrows 2008).

3.5 Conclusion

This study, as a part of the forest thinning trial in Wungong catchment to increase both water and environmental benefits, adds to our understanding of the jarrah forest ecosystem and its response to disturbances such as tree thinning. Results of the present study revealed that the short term (0-1 years) effects of thinning on the overstorey and mid-storey structure and composition of the jarrah forest were significant, but the effects on understorey species richness and ground cover were not significant. The latter effects were, however, under estimated in the log+notch thinning because sampling did not include the areas directly disturbed by logging, which comprised up to 10 % of the forest area in the present study. In this study, both the species composition and distribution of tree stands at different diameter size class have been altered by thinning which may have lasting effects on the structure and composition of the forest stands in the future. To explore the longer term impacts of thinning on structural components of the jarrah forest ecosystem, a series of further studies, focussing on both overstorey and understorey components, is essential. As fire is a core management practice in the forests of south-west Western Australia, the effects of thinning of the forest stands needs to be studied combined with periodic controlled burning.

Chapter 4: Dead woody debris resources in relation to thinning

4.1 Introduction

In contemporary forest management, the principle of sustainability is coupled with the principle of forest multi-functionality, which should be applied to all activities during sustainable forest management (Debeljak 2006). Effective implementation of the sustainability principle in forestry is dependent on the state of knowledge about the growth and development of a forest ecosystem, its structure and function.

Growth and development of Australian forest ecosystems can be described by the forest developmental stages identified in European temperate forest ecosystems (Leibundgut 1959, Korpel 1993). The forest passes through a sequence of phases of growth and ageing, and finally enters into the phase of breakdown of forest components due to natural ageing and superimposed anthropogenic disturbances such as logging and clearing. Both the natural and anthropogenic disturbances represent a destruction period (Holling 1986). This is the period when the forest ecosystem maximizes accumulation of dead woody materials in order to optimize adaptations to the prevailing environmental factors through building new ecological structures (Bobiec et al. 2000).

The terms coarse woody debris (CWD) and dead woody debris (DWD) have been used interchangeably in the literature (Pyle and Brown 1999, Grove 2001). Dead woody debris in this paper is used to demarcate all forms of dead woody materials such as fallen whole trees, logs, limbs and twigs, standing dead trees (also called snags or stags) and stumps that are ≥ 10 cm in diameter present in the forested study areas (Pyle and Brown 1999, Robertson and Bowser 1999, Fridman and Walheim 2000, Siitonen et al. 2000). In previous studies of dead wood, there has been considerable variation in sampling method, intensity, size of the plot, diameter of the dead wood recorded, and classification of recorded dead wood into different categories such as log, snag or stump and stages of decomposition. Moreover, there was considerable variation in the size threshold of the dead wood to be classified as fine and coarse debris depending upon the objectives of the study and the types of the forest ecosystem under examination (Harmon and Sexton 1996). From studies quantifying both fine and coarse DWD, the diameter threshold between fine litter and DWD has commonly been 1 cm (Turnbull and Madden 1986, Polglase and Attiwill 1992) or 2.5 cm (Moore et al. 1967, Woldendrop 2000), while some others recommended the size threshold between fine and coarse woody debris be set at 10 cm for the large end of a wood piece for most forests (Harmon and Sexton 1996).

Dead woody debris is an important structural component of many forest ecosystems, and plays an important role in a number of aspects of ecosystem functioning (Harmon et al. 1986). It is not only important in nutrient recycling (McCarthy and Bailey 1994) but also important for providing rooting substrate for plants (Harmon and Franklin 1989, McAlister 1995). It can provide sites for seedling establishment (Scott and Murphy 1987, Harmon and Franklin 1989, McKenny and Kirkpatrick 1999) and provide shelter and habitat for many types of vertebrates, insects, fungi and micro-organisms (Maser and Trappe 1984, Meggs 1996, Lindenmayer et al. 1999, Grove 2002, Grove and Meggs 2003). In aquatic systems, downed DWD plays an important role in maintaining stream quality and nutrients, especially by helping in sediment trapping and providing habitat for aquatic life (Van Sickle and Gregory 1990, Minore and Weatherly 1994, MacNally et al. 2002).

The DWD materials are the product of different external and internal factors that may have an effect on different scales from relatively large areas due to the influence of disturbances such as wind blow, ice break and fire and or on individual trees (e.g., senescence, diseases, and low vitality). The most important factors accounting for the origin and distribution of

DWD materials in managed forests are related to the human-induced activities, mainly forest management practices such as logging and fire (Harrod et al. 2009).

Despite the ecological importance of DWD in terms of biomass, structure and ecosystem functions, there is insufficient information about quantity and structural distribution of DWD materials in relation to forest management in the jarrah forest of Western Australia. However, a recent review of DWD in Australian forest ecosystems has cited a series of published and unpublished data from different forests types to provide an account of quantity, quality and other attributes of forest floor DWD (Woldendorp and Keenan 2005). This review emphasized that there are still a limited number of studies on quantity, quality, distribution and significance of DWD in shaping the overall structure and composition of forests in relation to forest management activities. Hence, structural attributes such as quantity, quality and stages of decomposition are of high significance for future studies of carbon and nutrient stocks and dynamics, fire risk assessment and its management, and to ensure sustainable forest ecosystem management. It is also of high importance in the study of forest biodiversity, for example, in wildlife habitat studies (Bate et al. 2004).

There are considerable differences in the quantity, structure and composition of DWD material from the studies for each forest type due to different stand structures, forest management and disturbance histories (see Woldendorp and Keenan, 2005). The use of differing methodologies, such as methods of DWD sampling and various size thresholds separating litter from DWD; are some of the factors affecting estimates of DWD resources within and among the forest types. There are very few studies directly examining the quantity, structure and composition of various components of DWD materials in relation to forest management activities such as thinning, regeneration cutting and prescribed burning (Grove 2001, Pedlar et al. 2002, Debeljak 2006, Motta et al. 2006); however, they have clearly reported the previous forest management histories of their respective study areas. As

DWD plays an important role in several aspects of forest ecosystem functioning, it would be wise to understand the impact of prescribed forest management activities on the DWD components of forest ecosystems. Study of DWD in relation to forest thinning is equally important because the ecology of the jarrah forests of south-west Western Australia has been described as fire-dependent, and most of the jarrah forests species are highly adapted to moderate to high intensity fire (Burrows 2008). An understanding of the dynamics of DWD resources for sustainable forest management is essential for formulating fire management policy. Clearly thinning has the potential to create a large mass of new DWD of diverse types that may be distinctively different to those produced by regeneration cutting alone or fire.

The main aim of this study was to quantify the amount and composition of DWD resources in a dry eucalypt forest of Western Australia in relation to different forest thinning practices.

4.2 Materials and methods

4.2.1 Study area and previous catchment management environment

Overall management of Wungong Catchment, along with climate, vegetation, and forest management practices, and the details of research design and thinning intensity, are described in Chapter 3.

4.2.2 Sampling design

Dead WD sampling locations were selected based on the earlier catchment management practices and the current thinning trial. Dead WD samples were surveyed in 9 plots of 90×70 m laid out in a 3×3 Latin square design for three treatments. All plots of 90×70 m were divided into three sub-plots of 70×30 m area to make the sampling easier. A fixed area plot (FAP) method was used for DWD sampling (Hingston et al. 1981, Turnbull and Madden 1986, Grove 2001). Dead WD was sampled in plots receiving three different treatments:

thinning followed by notching (log+notch), stem-injection of herbicide (notch-only), and a control (C) in the Vardi Road sub-catchment. Dead WD samples were assessed before and after the thinning treatment to assess treatment effects on post-thinning DWD resources.

4.2.3 Measuring DWD

In this study, methods were chosen that were as consistent as possible with other published and unpublished studies on DWD (e.g., (Hingston et al. 1981, Meggs 1996). The main problem was to ensure adequate size of the samples to be recorded for assessing this patchily-distributed resource. To overcome this problem, sampling plots comprised not less than 70 metres in length with a total width varying from 20 to 30 metres.

Three types of DWD were characterized: (1) logs that were either downed or leaning deadwood; (2) snags or stags (vertical deadwood > 1.5 m in height), and; (3) stumps (vertical deadwood ≤ 1.5 m in height). For each log, length and diameter at both ends was measured using tree calipers, and the mean diameter was calculated. Only pieces ≥ 10 cm in diameter (Pyle and Brown 1999, Fridman and Walheim 2000) and ≥ 50 cm long were considered. When applicable, lengths and diameters were taken at the point where the log either extended outside the plot boundaries, or tapered to ≤ 10 cm. For snags, the diameter at breast height (DBH) was recorded and heights were estimated using a digital hypsometer (FORESTOR VERTEX, Forestor AB Sweden). In the case of stumps, height was measured from the ground level and the diameter was measured at the top of the stump.

4.2.4 Decay stage

Each piece of DWD was assigned to one of the five decay classes for the logs as described by several authors (Thomas et al. 1979, Sollins 1982, Maser and Trappe 1984, Pyle and Brown 1999, Spetich et al. 1999, USDA 2001). Characteristics and properties of DWD used to classify the stage of log and snag decomposition are presented in Tables 7 and 8, respectively, with relevant pictorial illustrations in Figures 20 and 21, respectively.

Table 7 Characteristics of wood in five decay classes used for determining state of decay of forest floor logs and stumps based on Pyle and Brown (1999), Spetich et al. (1999) and USDA (2001).

Decay class	Log characteristics
1	Most of the bark present; branches retain twigs; solid and fresh wood; normally round shape; original color and log elevated on support points
2	Some bark may be present; twigs absent; decay beginning to occur but wood still solid and intact; normally round shape; original to slightly faded colour and logs elevated on support but sagging slightly
3	Bark generally absent or present as traces; twigs absent; more extensive decay but structurally sound; still round shape; moss, herbs and fungal bodies may be present; light faded colour; log sagging near ground and some termite damage in warm climates
4	Bark and twigs absent; early advance stage of decay; round to oval shape; when kicked log will cleave into pieces or can be crushed; partially solid but small blocky pieces; light brown to faded brown or yellow colour; moss, herbs and fungal bodies may be present; all of the log fully sagged on ground and extensive termite damage (in warm climate)
5	Bark and twigs absent, advance stage of decay; oval shape; soft and powdery when dry often just a mound; faded brown to yellow or grey colour; moss, herbs and fungal bodies may be present; all of the log fully sagged on ground

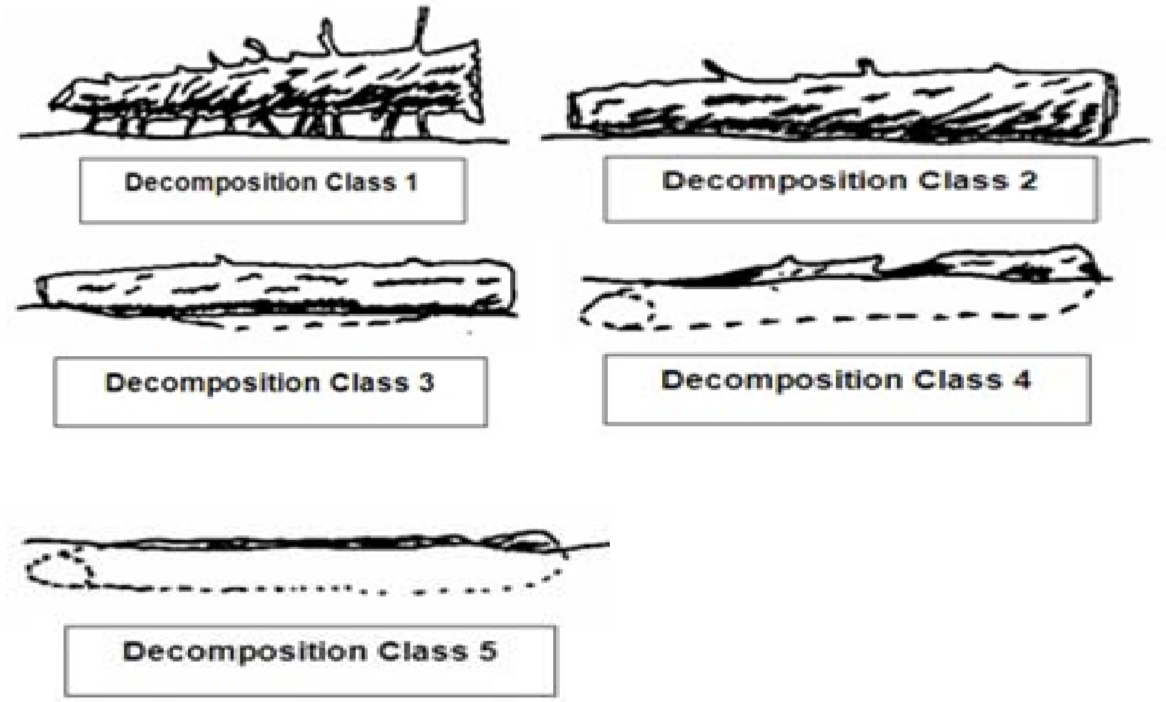


Figure 20 Log decomposition examples (see Table 7 for full descriptions of classes).

Table 8 Characteristics of wood in five decay classes used for determining state of decay of standing dead trees based on Thomas et al. (1979) and Sollins et al. (1987).

Decay class	Characteristics
1	Recently dead; branches and twigs present and bark intact and tight on bole
2	Bark loose and or partly absent; large branches present; much of the crown broken and bole still standing and firm
3	Large branch stubs may be present; top may have broken; bark generally absent; bole still standing but decayed and some termite damage
4	Branch and crown absent; bark absent; broken top; wood heavily decayed or hollowed and extensive termite damage producing hollows (soft wood < 70 %),
5	No bark; no twigs; highly broken top; wood is heavily decayed or hollowed; wood hard to soft or powdery (soft sapwood >70%)

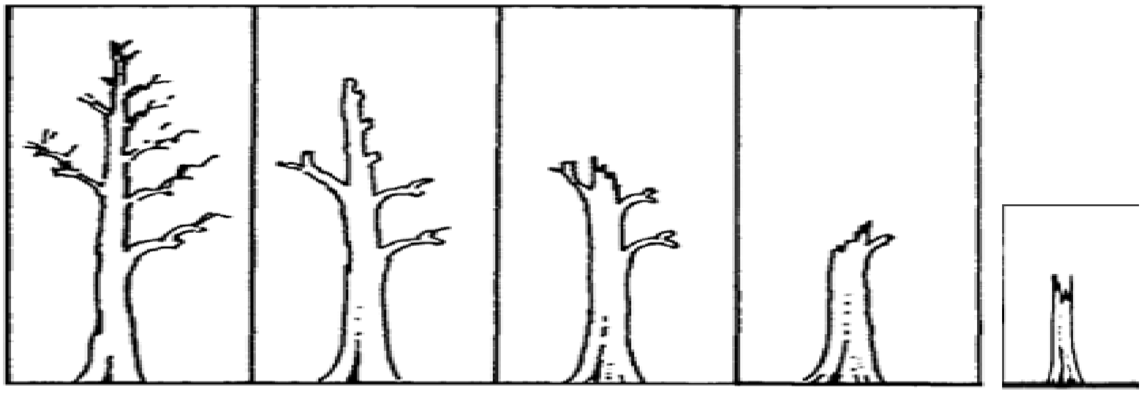


Figure 21 Snag decomposition classes (see Table 8 for full descriptions of classes).

4.2.5 Volume and mass calculation

For all the sites where the plots were used for sampling, volume estimates were calculated for each piece of DWD from its length and mean diameter assuming that all the wood pieces were cylindrical in shape. The DWD volume was calculated using the simple arithmetic formula $V = \pi r^2 l$ where V = volume of individual wood piece, r = radius of each piece calculated from its mean diameter, l = length and $\pi = 3.141592654$. Volumes of individual wood pieces were summed for each plot and the plot values were converted to a value per hectare.

For the calculation of dry mass of the DWD from its volume, 15 DWD pieces were collected from the site by cutting sections with a pruning saw (McKenzie et al. 2000), covering a full range of decay stages (Tables 7 and 8) comprising at least 3 samples for each decay class. Each sample was sealed in a plastic bag to preserve its original field moisture content and volume before transferring it to the laboratory. From each sample, sub-samples (size ranging from 74 to 215 cm^3/piece) along its cross-section were prepared. Fresh weight and volume of the wood pieces were measured for each sub-sample using the volumetric water displacement method (Stewart and Burrows 1994). Samples were wrapped with a thin plastic sheet before immersion to prevent porous samples from absorbing water (Stewart and Burrows 1994). Sub-samples were then oven dried at 72 ˚C to a constant weight. Total dry mass (TDM) was calculated by multiplying fresh volume by the ratio of dry weight to fresh volume of its sub-sample (Table 9).

4.2.6 Statistical analyses

Each variable derived from the dead wood data for each of the treatment plots was subjected to one-way analyses of variance (ANOVA) in the statistical package SPSS statistics 17.0 (SPSS 2008). Although the plots were laid out as a latin square design the data were analysed as a completely randomised design. To minimize the confounding effects of pre-thinning values on treatment effects on DWD volume, pre-thinning data were treated as covariates. To compare treatment effects, univariate ANOVA was employed. A general linear model was used for ANOVA. Type III sums of squares were employed as there were an equal number of replications in each treatment. The Bonferroni test was performed for pairwise comparisons amongst the treatments where significant differences were observed at $p<0.05$.

4.3 Results

Many variables showed significant differences at both plot and treatment level, indicating a high degree of heterogeneity among experimental treatment plots. Since the focus of this chapter is on the effects of treatment on DWD volume, pre-thinning differences were discussed only where co-variates were significant. Pre-thinning volumes were used as covariates to reduce the confounding effects of pre-thinning site differences on the post-thinning DWD parameters. Use of covariates in the model removes the effects of the relationship between the covariates and the dependent variable and provides a more precise estimate of the amount of variance by the factors in the model.

4.3.1 Extent and composition of DWD

There was a significant difference in the mean DWD volumes for the three treatment groups [F (3, 6) = 18.016, p = 0.004] (Table 9). The mean DWD volume comparison between thinning treatment types indicated that log+notch and notch-only treatments had significantly higher DWD than control treatments but were not significantly different from each other (Figure 22). There was a decrease in dry weight density of the DWD from decay class 1 to decay 5 (Table 10). Decay class 1 had the highest mean dry weight density (0.790 g cm^{-3}) while decay class 5 had the lowest mean dry weight density (0.419 g cm^{-3}).

Table 9 Significance levels for ANOVA with various attributes of dead woody debris (DWD) as the dependent variable at treatment level (L-log+notch, N- notch-only and C-control) for the dry eucalypt forest of south-west Western Australia. Pre-treatment values were used as a co-variate.

Variables	F value	Model probability value	Probability value for co-variate	Bonferroni test
DWD volume	18.02	0.004	0.230	L>C, N>C
Log volume	126.2	0.000	0.000	L>N
Snag volume	12.93	0.009	0.538	L>C, N>C
Stump volume	8755.4	0.000	0.000	L>C, L>N
Diameter size 10-30	6.21	0.039	0.256	L>C, N<C
Diameter size 30-50	5.529	0.048	0.534	N>C
Diameter size 50-70	3.858	ns	0.745	
Diameter size 70-90	6.408	0.036	0.662	
Diameter size >90	8.76	0.020	0.064	
Decay Class 1	9.247	0.018	0.642	L>C, N>C
Decay Class 2		ns		
Decay Class 3		ns		
Decay Class 4		ns		
Decay Class 5		ns		

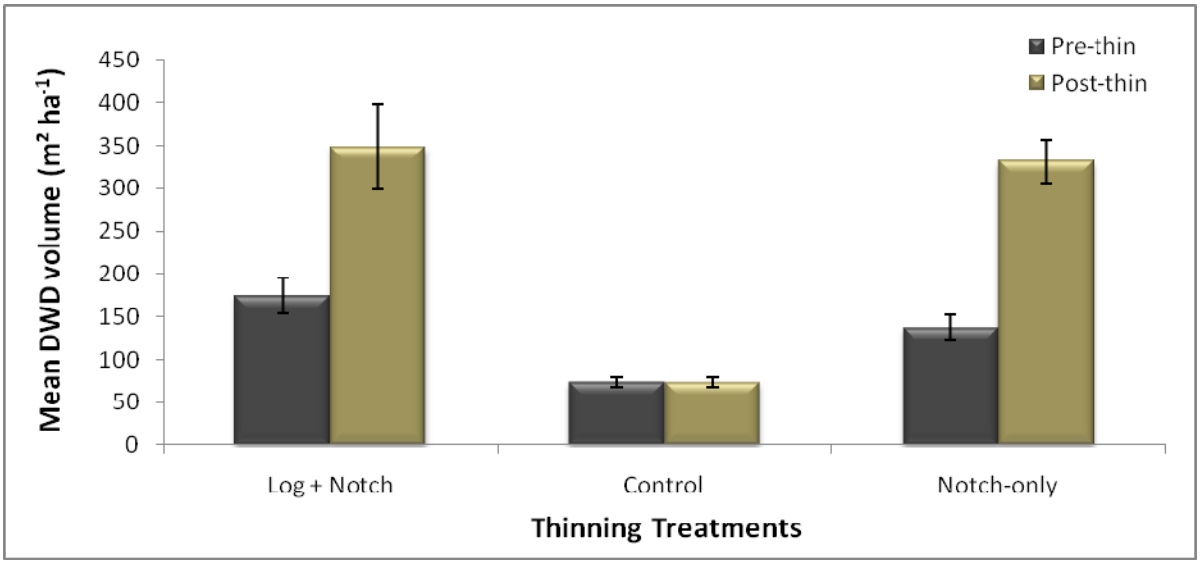


Figure 22 Pre- and post-thinning mean dead woody debris (DWD) volume ($m^3\ ha^{-1}$) for three thinning treatments. Values are means of 3 replicates. Vertical bars indicate standard errors. See Table 9 for significance of treatment differences.

Table 10 Dry weight density of dead woody debris samples ($g\ cm^{-3}$) from the dry sclerophyll forest of the south-west Western Australia by stage of decay class (see Table 7 for differences among decay classes). Samples were collected from Wungong catchment.

Decay Class	n	Minimum	Maximum	Mean	Std. Error
Decay Class 1	3	0.78	0.8	0.790	±0.0066
Decay Class 2	3	0.65	0.81	0.723	±0.0483
Decay Class 3	3	0.6	0.69	0.642	±0.0285
Decay Class 4	3	0.48	0.49	0.487	±0.0051
Decay Class 5	3	0.4	0.45	0.419	±0.0051

4.3.2 Composition of DWD volume by wood type

There were highly significant differences in the mean DWD volumes of logs, snags and stumps among the treatment groups ([F (3, 6) = 126.202, p = 0.000], [F (2, 6) = 12.932, p = 0.009] and [F (2, 6) = 8755.389, p = 0.000]). Pre- and post-thinning comparison in the mean DWD volume within the treatment revealed that there was no significant difference in log and stump volume for log+notch and notch-only treatments. However, significant differences in the mean snag volume were observed within the log+notch and notch-only treatments (Fig 23). Significant changes in the percentage contribution of log, snag and stump to total DWD were observed in the post-thinning DWD composition in both log+notch and notch-only treatment as compared to pre-thinning composition (Figure 24).

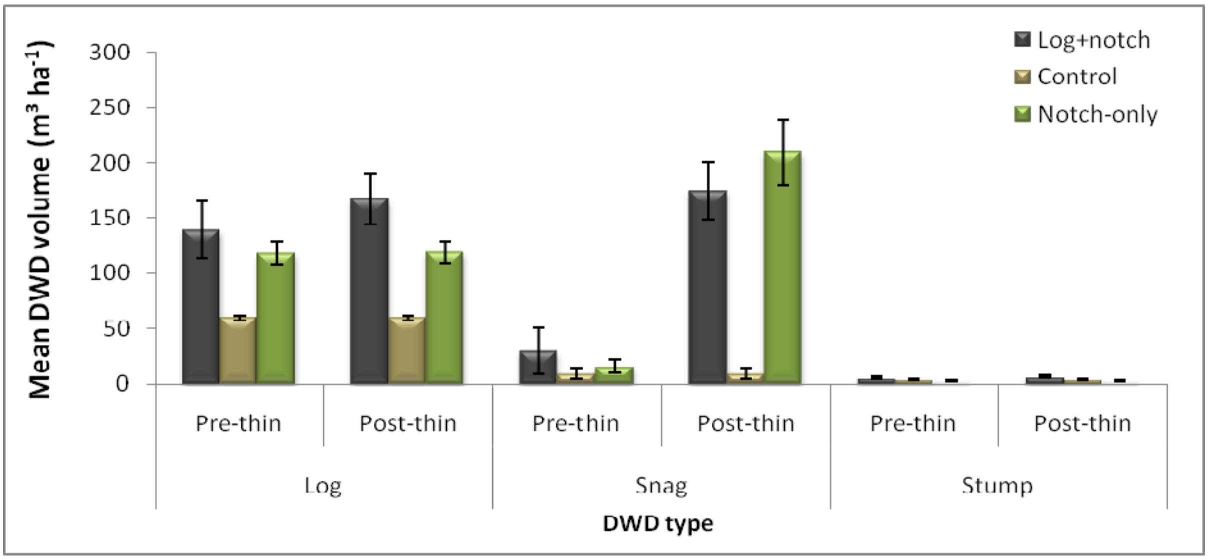


Figure 23 Mean dead woody debris (DWD) volume (m^3 ha^{-1}) by wood types for three different thinning treatments. Values are means of 3 replicates. Vertical bars indicate standard errors. See Table 9 for significance of treatment differences.

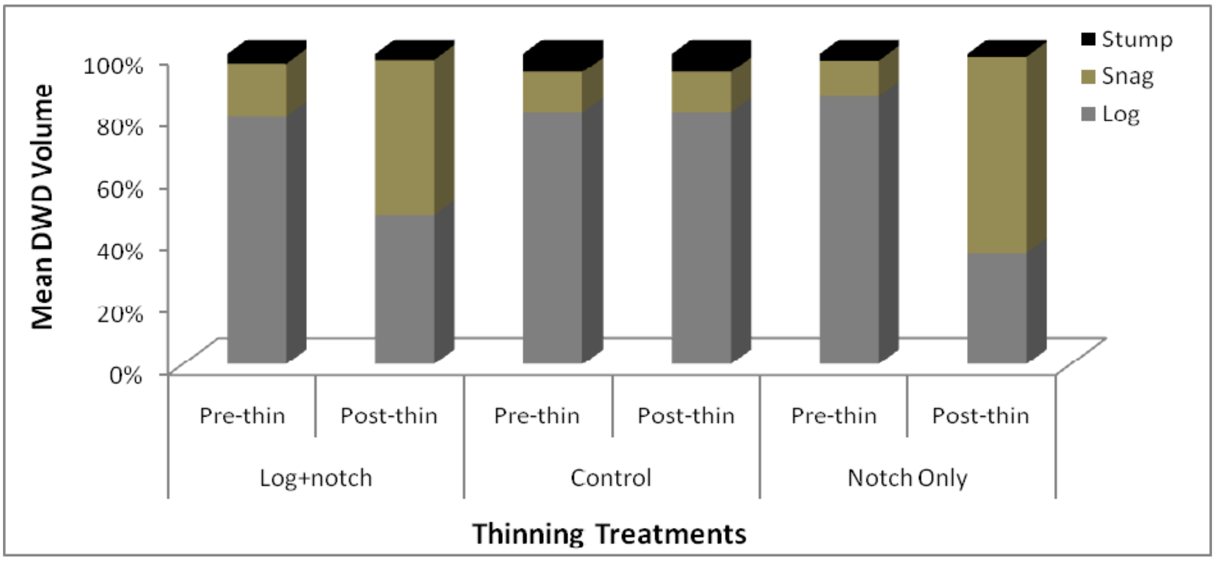


Figure 24 Proportion of total dead woody debris (DWD) volumes (%) by wood type. Proportions are means from three replicates.

The pre-thinning contributions of log, snag and stump materials to total DWD were approximately similar for all the treatment groups (80, 17 and 3 % for log+notch treatment, 81, 13 and 6 % for control treatment and 86, 11 and 2 % for notch-only treatment). Post-thinning composition of log, snag and stump were 48, 50 and 2 % for log+notch treatment and 36, 63 and 1 % for notch-only treatment (Figure 24).

4.3.3 Structural composition of DWD volume by diameter size class

Univariate analysis of variance indicated that there was a significant difference in the mean DWD volume among the treatments for all diameter size classes except for the 50-70 cm size class (Table 9). Pairwise comparison between the treatments for 10-30, 30-50, 50-70 and > 90 cm diameter size classes revealed that both the log+notch and notch-only treatments differed significantly from the control treatment but did not differ from each other (Figure 25). Comparison of the mean DWD volume in different diameter size classes showed a relatively normal distribution for both the pre- and post-thinning treatment conditions except for the increased proportion of DWD in the > 90 cm diameter size class of the thinned treatments (Figure 25). Before and after thinning differences in DWD volume within treatment for the same size class were greatest in the diameter size 30-50 cm with a mean of 56.2 m^3 ha^{-1} for log+notch and 63.1 m^3 ha^{-1} notch only treatments, followed by 10-30, 50-70, 70-90 and > 90 cm size classes, respectively (Figure 25). This indicates that the thinning of the tree stand was concentrated in the smaller diameter size classes with a progressive reduction of mean DWD volume for larger diameter size classes (Figure 25).

4.3.4 DWD volume composition by decay class

There was no significant difference in the mean DWD volume for all the decay classes except decay class 1. Mean DWD volume for decay class 1 was significantly increased by thinning treatments [$F (3, 5) = 9.247$; $p = 0.018$]. Pairwise comparison between the treatments for decay class 1 revealed that log+notch and notch-only treatments had significantly greater DWD than control treatment but did not differ from each other (Figure 26). In the pre-thinning condition, a large proportion of DWD was in a fairly early-advanced to advanced decay stage for all the thinning treatment plots. However, there was an abrupt increase in the mean DWD volume in decay class 1 as newly thinned dead wood entered in to decay class 1, hence the proportion of DWD in different decay classes had been changed (Figure 27).

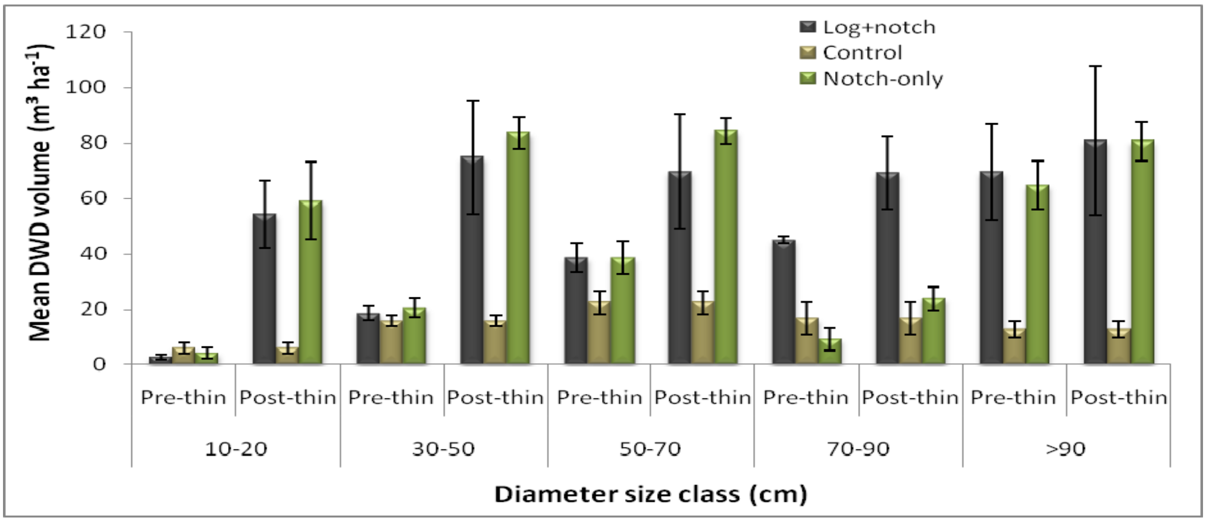


Figure 25 Mean DWD volume (m³ ha^{-1}) by diameter size class for three thinning treatments. Values are means of three replicates. Vertical bars indicate standard errors. See Table 9 for significance of treatment differences.

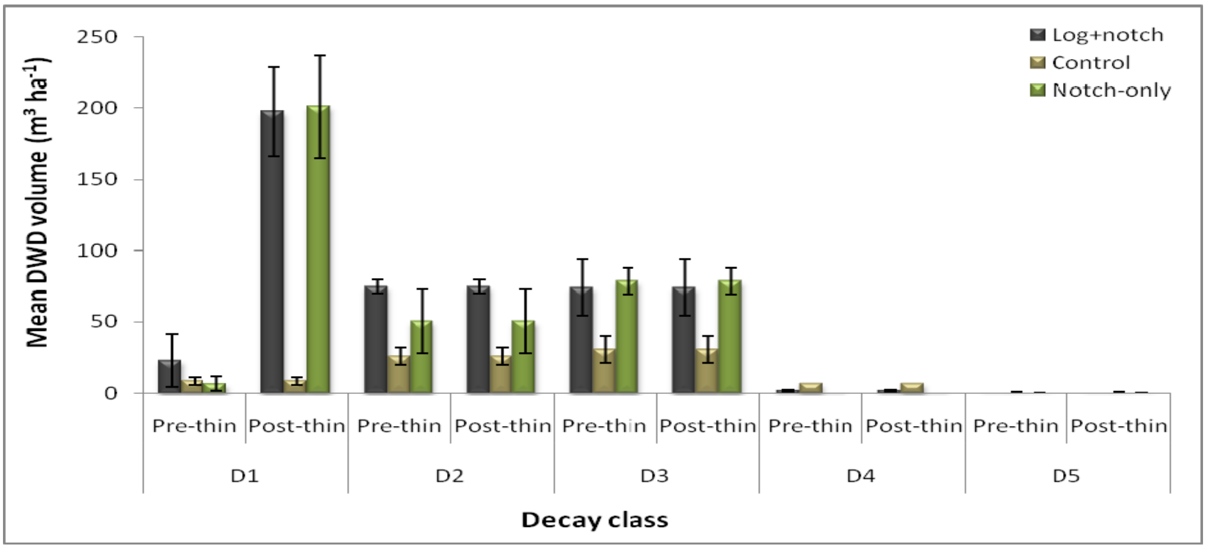


Figure 26 Mean DWD volume (m³ ha^{-1}) by decay class for three thinning treatments. Values are means of three replicates. Vertical bars indicate standard errors. See Table 7 for the description of decay classes. See Table 9 for significance of treatment differences.

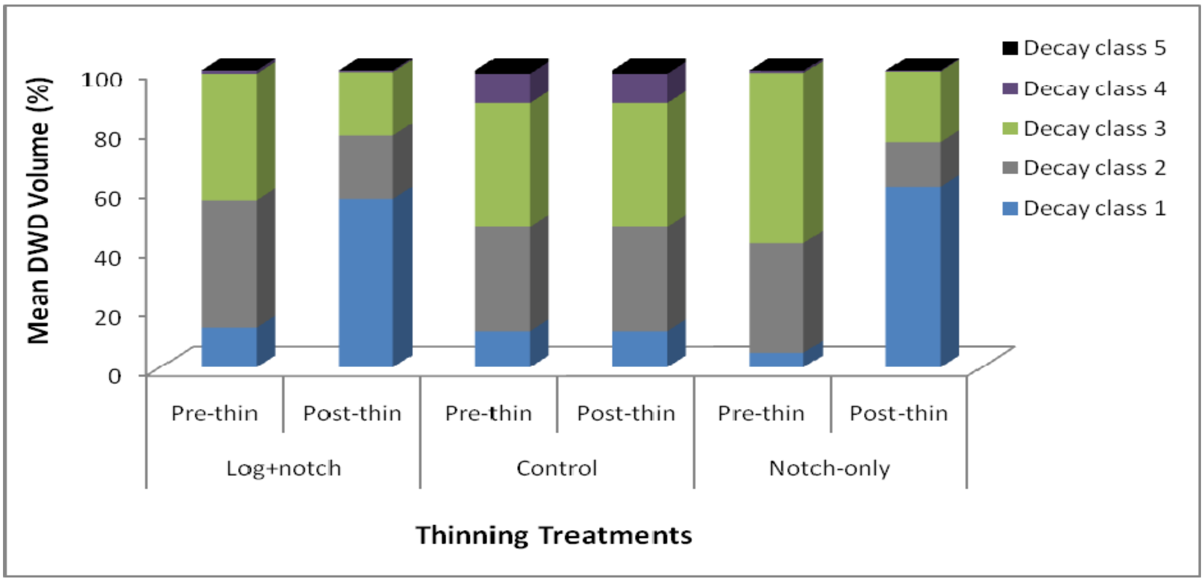


Figure 27 Proportion of total DWD volumes (%) by decay class. Proportions are means from three replicates. See Table 7 for the description of decay classes.

4.4 Discussion

4.4.1 Extent and composition of DWD

The DWD data gathered in this study, both volume and mass estimates, were unique for the jarrah-dominated dry sclerophyll forest ecosystem of south-west Western Australia. No previous study estimated the DWD volume to the level of logs, snags and stumps as components of the overall DWD pool in the jarrah forest ecosystem. There is a wide variation among estimates of DWD biomass on the forest floor in Australian forest ecosystems (Woldendorp and Keenan 2005). This might be due to the differences in the type of forests examined, varying degree of natural disturbances and management interventions and also due to the size thresholds used to differentiate fine from coarse DWD (Woldendorp and Keenan 2005). The use of differing sampling techniques and the time since last disturbance affect the extent and composition of DWD resources present in the forest ecosystem, making it difficult to compare estimates among forest types (Meggs 1996).

Mean DWD volume estimates (including logs, snags and stumps) in Australian forests were higher than the reported means for US temperate hardwood forests, but lower than the mean for US temperate coniferous forests (Harmon et al. 1986). Individual estimates in Australian forests ranged from 0.2 to 1089 t ha^{-1} (Turnbull and Madden 1986). The minimum individual estimate was in a 5-year-old *E. regnans* forest that regenerated after clear-felling and burning of logging residues (Polglase and Attiwill 1992), and the highest estimate was from a 63-year-old *E. obliqua* forest regenerated after fire (Turnbull and Madden 1986). Estimated DWD volume for this study as a mean for three different forest management scenarios was 251 m^3 ha^{-1}, (185 t ha^{-1}) which is within the range of 174-455 m^3 ha^{-1} for managed old-growth temperate evergreen forests of Tasmania (Meggs 1996). The present mass estimate (equivalent to 185 t ha^{-1}) appeared to be significantly higher than the DWD mass of 130 t ha^{-1} for the 60 year-old *E. marginata*- and *E. calophylla*-dominated forest of south-west Western Australia (Hingston et al. 1981), and significantly lower than the DWD mass of 1089 t ha^{-1} for the *E. oblique*-dominated Tasmanian forest (Turnbull and Madden 1986). In general, the values reported in this study for the dry eucalypt forest of south-west Western Australia tended to be higher than Australian lowland tropical rainforest (Grove 2001) and dry eucalypt forest of Western Australia (Hingston et al. 1981), but much lower than the managed old-growth temperate evergreen forest of Tasmania, Australia (Meggs 1996) and temperate evergreen *E. obliqua* forest regenerated from the fire (Turnbull and Madden 1986).

The mean DWD volume ranged from 62.4 m^3 ha^{-1} for the forest block with no thinning/logging disturbances in the last 60 years (control) to 332 and 349 m^3 ha^{-1} for the log+notch and notch-only treatments that recently received thinning interventions with 64.4 and 49.4 % basal area reduction, respectively (see Table 2). This large variation in the mean DWD volume between the unthinned and thinned plots was due to significant addition of fresh DWD volume generated after thinning to the DWD pool. Although there was a large difference in basal area reduction for log+notch and notch-only treatments, mean DWD volume in both the log+notch and notch-only treatments did not differ significantly. A total of 37.5 m^3 ha^{-1} log volume suitable for commercial use was removed from the log+notch treatment sites, thereby reducing the total DWD volume in the log+notch treatment plots. This suggests that volume estimates within a similar forest type differ significantly with different methods and intensity of forest thinning.

4.4.2 Composition of DWD volume by wood type

Of the three components of DWD studied (logs, snags and stumps), post-thinning log volume was generally highest in control and log+notch treatments followed by snags and stumps. However in the notch-only treatment, snags contributed a large proportion for the DWD pools, followed by logs and stumps. This difference in the snag volume after thinning in notch-only treatment can be attributed to the fact that individual trees were killed by stem-injection of the glyphosate herbicide in standing position, thereby increasing the snag volume in the total DWD pool. However, in most managed forests, the log component contributes most to the total DWD pool followed by snags and stumps (Grove 2001). A comprehensive estimate of logs, snags and stumps was presented for Rajhenavski Rog, Slovenian virgin and managed forest for four different forest cycle developmental phases (Debeljak 2006). In a comparative study of virgin and managed forests in Slovenia for optimal, mixed, regeneration and juvenile forest, log volume was higher than snag and stump volume in mixed, regeneration and juvenile forests whilst snag volume was higher than log and stump volume in optimal forest. The juvenile phase forest did not have significant snag volume, as the forest was only 4-5 metres high and had not reached the phase of natural senescence. Contribution of snags to the total DWD pool was minimal. This might possibly be due to cutting of dead trees from the forest while managing the forests.

In this study, a large amount of DWD in the form of snags might be of great importance for the ecological processes, such as rate of decomposition of the DWD components. Decomposition includes leaching, microbial mineralization and fragmentation of the DWD components (Brown et al. 1996). As most woods are high in polymeric material and low in soluble substrate, rates of mass loss and mineralization of nutrients from woody materials are expected to be slower as compared to the leaf litter components (Harmon et al. 1986). Rate of decomposition is highly influenced by several factors such as type (log, stump and snag), size and quality of substrate, prevailing climatic conditions (moisture and temperature) and the surface area to the volume ratios in contact with the soil surface (Brown et al. 1996). Hence, it is expected that the rate of decomposition is slower in snags as compared to logs and stumps due to lower surface area to volume ratios in contact with the soil (Brown et al. 1996). Brown et al. (1996) reported that the rates of DWD decomposition in Western Australian forest are relatively slow for major tree species, suggesting a relatively low importance of these resources as a supply of nutrients to the ecosystem. By contrast, DWD may be important as a carbon sink for up to many decades in the dry sclerophyll jarrah forest.

Dead WD provides habitat for many terrestrial vertebrates, including amphibians, reptiles, birds and mammals (Thomas et al. 1979, Grove and Meggs 2003). Thomas et al. (1979) identified 179 vertebrate species using DWD in the Blue Mountains of Oregon and Washington, which accounted for 57 % of the species breeding in that region. Several factors affecting the type and extent of animal use of DWD include physical orientation (vertical such as snags or horizontal such as logs), size (diameter and length), decay stage, species of DWD and overall abundance. Physical orientation of DWD is a major factor influencing vertebrate use. Birds and bats use snags whereas mammals other than bats use both logs and snags (Thomas et al. 1979). Thomas et al. (1979) reported that in the Blue Mountains of Oregon, only 20 % of the DWD-using species used both logs and snags. Additionally, most of the cavity-nesting birds (CNB) such as woodpeckers prefer snags with larger than average diameters aggregated in small patches for nesting (Bull 1975, Carey 1983). In this study, thinning of trees by notch-only treatment produced large numbers of snags in an aggregated patch that are expected to increase nesting sites for CNB in the future. The large volumes of logging residues produced by the log+notch treatment can be used as shelters by many animal species, including a wide range of small mammals, as logs provide protective cover (Thomas et al. 1979).

4.4.3 Composition of DWD volume by diameter size class

There were very few studies available that described the extent and composition of DWD in different diameter size classes (Grove 2001, Pedlar et al. 2002). The lack of consistency in classifying the total DWD volume into different diameter size classes made it difficult to compare among the studies. A recent study in boreal Canada for different forest types with varied disturbance history reported a distribution of total DWD volume into diameter size classes of 10-20, 20-30 and > 30 cm (Pedlar et al. 2002). Higher proportion of DWD volume was contributed by 20-30 and > 30 cm diameter classes, which appeared to be similar in this study as well. Proportions of DWD among different size classes in most of the forest types appears to be related to the diameter size-class distribution of the living trees (Grove 2001).

Significant differences were observed in the mean DWD volume among the treatments for different diameter size classes. However, there was no significant difference between log+notch and notch-only treatment at all diameter size classes except 70-90 cm. Field observation indicated that selection of large size trees for logging in the 70-90 cm size class in log+notch treatment largely contributed to the significant difference in the DWD volume between log+notch and notch-only treatments. Before and after comparison in mean DWD volume indicated that the difference among treatments was highest at diameter size class 30-

50cm followed by 10-30, 50-70, 70-90, and > 90 cm, respectively. These differences were directly related to the method and intensity of thinning operations because selection of trees during tree marking was targeted toward removal of small to medium diameter size trees except for the trees marked for logging.

4.4.4 Composition of DWD volume by decay class

There were no significant differences in the mean DWD volume between the treatment plots before thinning intervention. This indicated that if the forests were left unlogged for a 60-year period, composition of DWD resources would follow a natural pattern of DWD distribution across different decay classes as was observed in many forest ecosystems (Pedlar et al. 2002). In a study of boreal Canada, DWD was classified into three decay classes, fresh, moderate and rotten, with a higher proportion of fresh DWD for all the burnt and clear-cut for conifer, deciduous and mixed forests studied (Pedlar et al. 2002). The rotten class contributed little to the DWD volume in all the forest types except in burnt forest. Results of the present study (Figure 26) appeared to be consistent with the results of boreal Canada as the majority of pre-thinning DWD was in the decay classes 1, 2 and 3. The present result is consistent with the exclusion of fire from the study site for 10 years (Frank Batini, personal communication). Proportion of DWD volume in decay classes 1-5 for all the pre-treatment plots were 10, 38, 47, 4 and 1 %, respectively. Results of this study were further supported by the study conducted in an Australian lowland tropical rainforest where DWD contribution of each decay class to total DWD composition was 4, 12, 25, 43, and 20 % for decay classes 1-5, respectively (Grove 2001). Dead woody debris after thinning as compared to pre-thinning appeared to be significantly higher only in the decay class 1, as all the added DWD from thinning entered into decay class 1 based on the classification system used in the present study. It is assumed that decomposition of DWD starts as soon as it enters into the DWD pools. At this stage it is unclear how long DWD material in each decay class will stay in that class, as there is no previous Australian literature on it. However, changes in mass and nutrients in experimental logs of six tree species viz, *E. diversicolor, E. marginata, Pinus pinaster, Allocasuarina fraseriana, B. grandis* and *E. calophylla* during 5 years of exposure in the three major forest production regions of south-west Western Australia were measured to determine how climate, substrate quality and substrate size interact to regulate decomposition of woody debris in Mediterranean-type climate (Brown et al. 1996). *E. calophylla* lost the most mass during this time, up to 65 % of the initial mass, where as decomposition was least in *P. pinaster* and *E. marginata*, at about 24-26% of the original mass. Mass losses were greatest in Manjimup, the wettest site and least in Gnangara, the driest site, but the differences in overall level of decomposition were small despite the range in climatic moisture regimes. Based on the changes in density of DWD obtained in the present study (Table 10), these results suggest that woody debris of *E. calophylla* decomposed from Class 1 to Class 5 in only 5 years. By contrast the *E. marginata* debris decomposed from Class 1 to Class 3 in the same time. However, more detailed studies in the thinned forests would be needed to determine the actual rates of DWD decomposition. Such information is particularly important for estimating C stocks, and nutrient turnover, apart from impacts on wildlife habitat.

Dead WD in the jarrah forest of Western Australia has been expected to have great value as faunal habitat, particularly for invertebrates, but also for mammal, reptiles and birds (Meggs 1996, Grove 2002, Lindenmayer et al. 2002). Standing dead trees are critical foraging and nesting sites for many birds species in the temperate-zone forests (Hunter 1990). Abundance of the number of birds and bird species were significantly greater in thinned sites compared with unthinned sites in a Victorian eucalyptus forest (Barr et al. 2011). This may be attributed to the increased numbers of snags for nesting and foraging sites. Significant increase in the abundance, amount and structure of DWD in this study are expected to increase the numbers of cavity nesters as in northern forests (Niemi and Hanowski 1984, Douglas and David 1987). However, woodpeckers abundance followed a 'law of diminishing returns' where, beyond a certain point, increasing the numbers of trees of the types used did not lead to increased woodpeckers abundance (Gunn and Hagan 2000). Similarly DWD can play an important role in the storage and cycling of carbon in the forests (Brown et al. 1996, O'Connell 1997, Berg 2000). A better understanding of amounts and dynamics of these components of carbon stocks is required for carbon accounting and assessing the impacts of forest management practices on greenhouse gas balances. Furthermore, understanding of DWD amount, its size distribution and other fire related attributes are of high importance for fire managers for planning and implementation of fuel load reduction treatment in the forest so that the possibility of occurrence of wild fire can be minimized.

4.5 Conclusion

Improved knowledge of the extent and composition of DWD resources in forest ecosystems is important for sustainable management of forests in Western Australia as well as many other parts of the world. Knowledge of DWD is equally important in carbon budgeting in the forest ecosystem as DWD is a long-term source of stable carbon. In this study, no attempts were made to estimate the amount of carbon stored in the DWD resources present in the forest. Several studies have considered DWD as an important forest resource, but there is no consistency in the ways they deal with it. This study as a part of a forest thinning trial in the Wungong catchment within the dry eucalypt forest of south-west Western Australia used an integrated approach to generate key knowledge on extent, structure and composition of DWD resources of the jarrah forest in relation to different forest management scenarios. Results of the study indicated that the extent and structural composition of DWD resources largely depends upon the previous forest management activities such as logging and burning history, age and structure of the forest stand and more particularly to the methods and intensity of the thinning operation. Thinning significantly increased the quantity of DWD for both log+notch and notch-only treatments. It also changed the structure and composition of DWD components (i.e. log, snag and stump) by altering the percentage contribution of each component to the total DWD pool. Changes in the structure of the dead woody resources after thinning revealed that the amount, structure and composition of DWD in the forested catchment was directly related to stand age and structure of the living trees and more specifically to the method and intensity of forest thinning. Hence, the knowledge on extent, structure and composition of DWD resources will be of high importance for the forest management authorities in planning for multiple uses of forest resources.

Chapter 5: General discussion

Based on the research findings hypothesis 1 and 3 were accepted while hypothesis 2 was rejected. Log+notch and notch-only treatments significantly altered forest structure according to the prescribed decrease in stand density and basal area of smaller, high-water use trees. Similarly, both the treatments led to a significant increase in the amount of DWD and its distribution among types. Effects on understorey ground cover and species richness following log+notch and notch-only treatments were not significant after one year.

A major issue faced by jarrah forest catchment managers in south-west Western Australia is how forest management practices relate to the quantity and quality of water output produced from the catchment and their effects on jarrah forest ecosystems, especially on vegetation structure and composition. The contributions of different land-use and catchment management practices to catchment water yield in the jarrah forest of south-west Western Australia have been well studied (Ruprecht et al. 1991, Ruprecht and Stoneman 1993, Roberts et al. 2000, Bari and Ruprecht 2003), but comparatively less is known about the impacts of forest management practices, especially prescribed thinning, on the vegetation component of the jarrah forest ecosystem. Therefore, the present study was conducted to understand the short-term impacts of forest thinning treatment on vegetation structure and composition of the Wungong catchment of south-west Western Australia, which has a Mediterranean-type climate (cool wet winter and dry summer).

The strongest influence on the composition of the ground flora in an even-aged re-growth forest is the nature and biomass of the overstorey (Metzger and Schultz 1984, Alaback and Herman 1988). The forest overstorey influences light penetration (Hill 1978), soil moisture, litter fall, root competition and soil moisture (Dryness et al. 1988). Thinning influences key forest structuring processes, including crown dominance and spatial heterogeneity (Carey

2003). In this study, methods and intensity of thinning exhibited significant impact on post-thinning stand density and basal area and their structural composition as compared to the pre-thinning forest stand or control (unthinned) stands (Chapter 3). Thinning treatment significantly decreased the stand density and basal area of the study plots: it reduced the basal area by 64.4 % for the log+notch treatment and 49.4 % for notch-only treatment. The greatest impact was observed in the small to medium diameter class trees (most of which were classified as midstorey) that received a non-commercial thinning. Significant reduction was also observed in the upper storey with large diameter class trees, which were subjected to commercial logging in the log+notch thinning prescription. Commercial thinning followed by non-commercial thinning effectively reduces the tree stocking to a level that, according to the local forest manager, would occur in the following two decades through natural self-thinning when the less vigorous trees in the stand are becoming sub-dominant and suppressed (Opie et al. 1984). However, young jarrah trees (specially in the bauxite mine rehabilitated sites) grow with little evidence of self-thinning (Ward and Koch 1995). While the short-term thinning effects reported here in terms of stand density reduction and changes in species composition are consistent with other studies in jarrah forest, the understorey species richness and ground cover responses are less easily compared with the existing literature, which is dominated by studies in northern hemisphere even-aged stands (e.g. Wienk et al. 2004; Metzger and Schultz 1984).

Natural disturbances are important drivers of several studied Western Australian ecosystems, and play a significant role in recruitment and regeneration of several eucalypts species (Burrows et al. 1990, Yates et al. 1994). Landscape-scale natural disturbances such as fires, floods and severe windstorms are some recurring events in the south-west Western Australian landscapes (Fitzpatrick 1970, Minor et al. 1980, Lourenz 1981, Burrows et al. 1990). Although most of these disturbance events occur infrequently, they may be amongst the most important processes driving the dynamics of native ecosystems in the Western Australian landscapes (Main 1987). Human induced activities such as clearing of native vegetation for agriculture has resulted in the transformation of natural ecosystems to predominantly agricultural land use, producing a new landscape of agricultural land, salt pans and native vegetation remnants (Saunders et al. 1993). Indeed, these all have been reported to be the case for many ecosystems elsewhere. The literature on the physical disturbance of ecosystems by intermittent events is vast and has been reviewed by several authors in Pickett and White (1985) and more recently by Attiwill (1994). For many plant species, disturbance provides the conditions for seedling recruitment (Pickett and White 1985, Attiwill 1994). However, success is dependent upon the co-occurrence of favorable rainfall in the first summer following seed germination. As a consequence, not all disturbance events result in recruitment. The extent to which such favourable conditions are necessary for particular outcomes in vegetation structure and diversity following thinning is not clear. That is, it remains unclear how closely recruitment and vegetation patchiness in thinned forests follows the patterns of disturbance ecology reviewed by previous authors (e.g. Attiwill 1994). There may well be differences between notch-only compared to log+notch since the latter involves greater physical disturbances. Longer term studies are needed to ascertain whether thinning in the jarrah forest should be treated as a disturbance event with ecological outcomes comparable to those obtained with disturbances like wind, fires etc. (e.g. Burrows et al. 1990; Yates et al. 1994).

Fire is the main disturbance factor affecting the vegetation dynamics of south-west Australian forests. Although the pre-European fire regimes are not well understood, there is evidence of fire occurrence in south-west Western Australian ecosystems that pre-dates the arrival of humans by thousands of years (Hassel and Dodson 2003) and evidence of regular burnings of the forest by Nyoongar Aborigines (Hallam 1975, Abbott 2003, Lamont et al. 2003).

Consequently, the vegetation displays the fire adaptive traits typical of fire-prone environments in other parts of the world, such as the ability to resprout after fire, serotiny, development of thick bark and enhanced germination by heat and smoke (Christensen and Kimber 1975, Bell et al. 1989, Burrows and Wardell-Johnson 2003). Both jarrah and marri, the dominant tree species of the jarrah forest, are well adapted to fire having the ability to readily resprout from below ground (lignotuber) or above ground (epicormic buds) (Burrows and Wardell-Johnson 2003). Burrows and Wardell-Johnson (2003) reported that small-stemmed trees develop thick, protective bark relatively quickly, while some suppressed and sub-suppressed trees did not show a measurable increase in bark thickness (Burrows et al. 2010).

A study conducted by Burrows et al. (2010) in a dry sclerophyll forest dominated by jarrah and marri, in the Perup forest of south-west Western Australia, reported that the mean stem growth measured over 20 years was lowest in the long unburnt treatment compared with the burn treatments, although surface soil nutrient levels were generally higher in the unburnt treatments, suggesting these sites may be moisture limited (Burrows et al. 2010). No clear patterns were observed of the effects of the burn treatments; including the number of fires and the interval between fires on tree stem growth, stand basal area increment, crown health or mortality.

Several factors could initiate and maintain the patchiness and composition of populations of *E. marginata*, *E. calophylla* and *B. grandis* in the jarrah forest. Primarily, these include micro-environmental heterogeneity, interspecific competition, and physiological requirements. There is little information available for evaluating all these factors; however, interspecific competition appears to be more important (Abbott 1984). In the low rainfall areas east of the zone of high quality jarrah forest (Abbott and Loneragan 1983), *B. grandis* becomes infrequent and eventually disappears, probably because soil moisture storage is insufficient to support both *B. grandis* and *E. marginata* (Havel 1975b). Furthermore, inability of the *B. grandis* seeds to disperse more than 20 metres from the seed tree by wind may contribute to the patchiness of the *B. grandis* in the study area (Abbott, unpublished). It is also suggested that whatever the kind of disturbance that occurs, if a supply of *B. grandis* seeds is unavailable, obviously there is no response to disturbance (Abbott 1984). In the present study, the population of *B. grandis* appeared to be lower than *E. marginata* and *E. calophylla* at the treatment level; however, the numbers were substantial at the plot level (plot 1, 6 and 9). Increased relative abundance of banksia after thinning may be expected to enhance its post-thinning recruitment success in the short term. Over time, greater recruitment of banksia may also increase its density in the forest and hence the level of seed dispersal. The extent to which this leads to a gradual shift towards greater banksia dominance in the thinned forest may depend on the presence or absence of *Phytophthora cinnamomi* and the fire regime.

The structure and reproduction of any forest are closely related because the methods of reproduction largely determine the structure of the forest (Jones 1945). All tree species in the jarrah forest exhibit a wide range of diameter classes, typically reflecting a wide range of species characteristics and age classes. Species in the different stands often show notable differences in the frequency of diameter-classes (Abbott 1984). Thus, there is no single diameter-class structure for all three tree species and the tree diameter size distributions were significantly variable. These facts considered together point to the populations of tree species in the jarrah forest consisting of groups of even-aged plants, with any stand consisting of a collection of such groups of trees recruited at different times (Stoate 1923).

In this study, differences in the short-term response of the understorey vegetation to the two thinning methods and intensities implemented in the Wungong catchment of the jarrah forest were of less significance floristically than the difference due to pre-thinning local site variability. Differences in species richness within the treatments before and after thinning were not significant in all the treatments. However, the species recorded differed between the treatment groups. This difference in the species recorded is likely related to the site variability between the treatment groups. One year of monitoring understorey vegetation following thinning did not show significant effects on the understorey species richness and ground cover. This may be attributed to the minimal site disturbance during thinning and also to the decreased availability of available soil moisture due to dramatic decrease in the mean annual rainfall after thinning. An increase in thinning intensity can lead to a wide variety of vegetation changes, including an increase in ground fern cover as in Picea-Tsuga forest of boreal Canada (Alaback and Herman 1988), a reduction in woody species as in Haldimand-Norfolk region of southern Ontario (Reader 1988), an increase in woody species (Wetzel and Burgess 2001), or a reduced cover of the herbs and an increase in vines and broadleaved species in south Arkansas and north Louisiana (Shelton and Murphy 1994). Alternatively, an increase in thinning intensity may not affect species richness and composition (Locasio et al. 1990), or produce highly variable response (Alaback and Herman 1988, Puettmann and Berger 2006). Where compositional differences due to different thinning treatment intensities are reported, the responsible mechanisms have included differential shade tolerances and the competitive interaction of woody and herbaceous species promoting shade tolerant species (Crawford 1976), and surface soil disturbance (Reader 1988). However, floristic change following thinning was reported to be minimal in the absence of soil disturbance (Brunet et al. 1996). Unfortunately, the placement of quadrats in the log+notch plots avoided areas of direct disturbance by logging machinery, so the potential for understorey composition change was not directly assessed. Nevertheless, the notch-only thinning may be expected to cause less change in the longer term on understorey composition than log+notch. Minimizing the extent of physical disturbance during logging in the log+notch thinning prescription may limit the risk of understorey composition change after thinning. However, in any case the extent of disturbance in the log+notch treatment would be similar to or less than with existing commercial logging operations in the forest.

Forest management activities such as thinning and burning significantly affect the amounts of DWD in a forest (Harmon and Hua 1991), so a considerable difference in the estimates of DWD were observed in the review of coarse woody debris in Australian forest ecosystems (Woldendorp and Keenan 2005). Thinning of the forest stands in this study significantly increased the amount of DWD in the forest (Chapter 4); however, the form and structure of DWD are largely dependent upon method and intensity of thinning. Logging residues left on site can increase dead biomass in the forest floor DWD pools (Harmon et al. 1986). For example, a southern Tasmanian *E. regnans/E. obliqua* forest that had been clear-felled 36 years previously had approximately 616 t ha^{-1} of forest floor DWD > 15 cm in diameter, mostly as logging residue (Woldendorp et al. 2002). However, in those forests, reducing standing biomass to a small fraction of the amount found in the natural forest stands reduces the potential future volumes of the forest floor DWD (Hodge and Peterken 1998) and snag formation (Cline et al. 1980).

In the present study, no attempts were made to estimate the rate of decomposition of DWD of the jarrah forests. Nevertheless, the quantity of DWD volumes at different stages of decomposition were estimated following the classification system of Pyle and Brown (1999), Spetich et al. (1999) and USDA (2001).

In conclusion, the research presented in this thesis has documented the short-term impacts of forest thinning treatment on stand structure and composition. It demonstrated that thinning can significantly change the structure and composition of overstorey components of the jarrah forest as well as the DWD component that is expected to have a long-term impact in the forest. Minimal changes in the understorey components in response to overstorey thinning may be attributed to the insignificant physical disturbance in the quadrats assessed and perhaps to the short time frame of this investigation. Further study of the response of understorey species richness and ground cover in the longer-term will be important in understanding the post-thinning ecological processes.

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